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by

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EDITORIAL

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Protect remaining roadless areas: The IENE 2014 declaration

IENE[1]

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Introduction

Today's transport systems with their supporting infrastructure entail substantial environmental impacts and consume non-renewable natural resources including our most limited asset: space. With ever more roads and railways being built or upgraded, further encroachment on and disturbance of natural living spaces is inevitable (Laurance et al. 2014). Much of this development takes place in developing countries, but also in Europe, and especially in eastern countries, new transport infrastructures lead to significant loss and fragmentation of habitat (Jaeger et al. 2011). Although the consequent threats to biodiversity are well understood, existing tools and strategies to prevent further fragmentation and continued loss of biodiversity appear not sufficiently effective (European Commission 2010). Even in the Impact Assessment of the European White Paper on Transportation (European Commission 2011), habitat fragmentation is merely recognized but not solved. In response, European countries engage in developing a green infrastructure of networks of habitats that shall re-linking nature protection sites across fragmented landscapes (European Commission 2013). Transportation corridors are major obstacles to this endeavour, but they can be overcome and even partly integrated in the design of green infrastructure (IENE 2012).

In fact, many of the direct and immediate threats of roads and railroads can be partly overcome or at least minimised (Iuell et al. 2003; van der Ree et al. 2015). It is also evident, however, that not all negative impacts can be mitigated or compensated for. There will always be a residual detrimental effect on nature. It is therefore only logical to conclude that in certain areas, where these residual effects are not acceptable, construction of transport facilities should be entirely avoided. Such areas need to remain (or become again) roadless to provide sufficient undisturbed space for nature conservation (DeVelice and Martin 2001, Crist et al. 2005). Even within Europe, where only small and few roadless areas reside (Selva et al. 2011, 2015), this need is increasingly recognised.

To support this development and highlight the value of roadless areas as complements to current biodiversity conservation in Europe, the participants at the IENE 2014 international conference (Seiler 2014) unequivocally asked for a pan-European strategy on roadless areas.

The following text is copied from the IENE 2014 website (http://iene2014.iene.info/iene-2014-declaration/):

The IENE 2014 declaration: Protect remaining roadless areas

"We, the participants of the IENE 2014 International Conference on Ecology and Transportation, acknowledge that:

- *the mobility of people and goods is important for economic development; transportation facilities such as roads, railroads and canals bring benefit to people and are essential components of present-day human societies,*
- *transportation infrastructure with its associated traffic exerts substantial pressures on biodiversity that extend far from individual transportation corridors and may interact and even accumulate at network level,*
- *even minor infrastructure is of significance as it prepares for exploitation of natural resources and secondary development,*
- *the detrimental environmental impacts of traffic and transportation infrastructure can only in part be mitigated effectively, but not entirely avoided.*

Roadless areas are of particular importance for biodiversity conservation, because they

- *are the least disturbed natural areas in the world,*
- *are characterized by high ecological value, integrity and connectivity,*
- *act as refuges for native and endangered wild animals and plants,*
- *provide vital ecosystem services such as clean water and air, opportunities for recreation, and protection against pests and invasive species,*
- *are more resistant to and resilient from catastrophic events,*
- *help species to adapt to new conditions created by climate and landscape change.*

Thus, roadless areas far exceed roaded areas in the ecological benefits they provide.

Europe has been fragmented by transportation infrastructure for a long time. Accordingly, preserving the continent's last remaining roadless areas will significantly contribute to prevent further loss of biodiversity. Preserving roadless areas is hence necessary for reaching the UN Aichi strategic goals and EU biodiversity targets.

Therefore we, the participants of the IENE 2014 International Conference, call for a pan-European strategy to protect roadless areas.

We urge that such areas are given a stronger conservation status in policy, planning and practice, both nationally and internationally, by

- *mapping and monitoring roadless areas at national as well as European level,*
- *incorporating roadless areas explicitly as conservation targets in national and European policy and legislation,*
- *avoiding infrastructure development in roadless areas,*
- *identifying areas of particular value for restoration as roadless areas,*
- *regularly monitor and evaluate the efforts to protect roadless areas,*
- *re-creating roadless areas by means of road closure and removal.*

The IENE 2014 International Conference has highlighted the ecological and social benefits of roadless areas, outlined solutions for how transportation infrastructure can be developed without compromising these benefits, and shown that the transport sector is able and willing to contribute substantially to implementing these solutions."

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EDITORIAL

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Greener transport infrastructure – IENE 2014 International Conference

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Introduction

Transportation and infrastructure are recognised as significant drivers in the global loss of biodiversity. Their impacts on nature are well described (Forman et al. 2003, Davenport and Davenport 2006) and there is ample evidence for the negative effects of traffic and transportation infrastructure on nature. Even though roads and railroads may occupy small areas compared to e.g. forestry and agriculture, their ecological effects may reach a large portion of the landscape (Forman 2000, Benítez-López et al. 2010), cause the death of millions of wild animals (Seiler and Helldin 2006), and disturb surrounding habitats through pollution with chemicals (Stengel et al. 2006) and noise (Barber et al. 2010) and the spread of alien species (Vilà and Ibáñez 2011).

The overall impact of transport infrastructure on nature is evident, but there are means to minimise the pressure, to adjust infrastructure facilities and, to some degree, introduce beneficial services for wildlife (Forman et al. 2003, Iuell et al. 2003). Such measures can and should be implemented as a standard in infrastructure development and maintenance. However, knowledge about the functionality and efficacy of such measures is not always satisfying. Technical innovations and new mitigation concepts need to be tested and evaluated. In addition, their functionality and effectiveness also depends on the interplay between the transport sector and other sectors of society. Communication, knowledge transfer, and public education are therefore just as essential as are legal frameworks, policies, technical development and environmental science. Current international policies in the field of nature conservation, such as the Aichi Biodiversity Targets under the UN Convention on Biological Diversity (UNEP 2010) and the EU-wide strategy on Green Infrastructure (European Commission 2013), are developing clearly in this direction, recognizing the transport sector and transportation facilities as important players in the endeavour towards a greener and sustainable future. Obviously, this calls for international collaboration in research and practice, for enhanced exchange of knowledge between disciplines, and for the development of harmonised standards and procedures that can be referred to by international actors.

To meet these demands, communities of practice have formed in several parts of the world (Wagner and Seiler 2015), such as the Australasian network for ecology and transportation (ANET, www.ecoltrans.net) in the Australasian region, the International Conference on Ecology and Transportation (ICOET, www.icoet.net) in North America, and the Infra Eco Network Europe (IENE, www.iene.info) in Europe.

IENE (Infra Eco Network Europe)

The Infra Eco Network Europe is a formalized network of mainly, but not exclusively, European authorities, institutes and individual experts working with the impacts of transport and infrastructure on nature and their mitigation (Spindler et al. 2014). Since 1996, IENE addresses decision makers, planners and researchers as well as the general public, and operates as an international and interdisciplinary arena to encourage and enable cross-boundary cooperation in research, mitigation and planning.

IENE national workshops and biannual international conferences on ecology and transportation provide recurring interdisciplinary forums for these activities. The conferences aim at presenting cutting-edge research, identifying urgent questions and problems, discussing effective solutions, and outlining the paths for upcoming activities in transport and infrastructure ecology.

The IENE 2014 International Conference brought together more than 200 professionals in the field of transportation, infrastructure and ecology, from 35 countries worldwide. With over 130 presentations and 6 workshops, the conference addressed the global ambition to achieve a "greener" and more ecologically sustainable transport infrastructure. Greener infrastructure stands for transport facilities that are well adapted to the ecological conditions of the surrounding landscape. The ambition for a greener infrastructure is expressed through striving for a wiser use of marginal infrastructure habitats to favour biodiversity and ecosystem services, for reduced disturbance and pollution by traffic, and for more permeable transport corridors that are safe for both humans and wildlife, and while acknowledging that not all impacts can be avoided and that certain areas must be kept roadless.

About this issue

This thematic issue of Nature Conservation compiles a selection of papers from the IENE 2014 conference. The following papers constitute a sample of the width of topics addressed in conference presentations and workshops, all aiming at providing guidance for management and conservation (see Seiler 2014 for all conference abstracts).

Many contributions to the conference dealt with the immediate conflict between traffic and wildlife. The presentations covered topics such as traffic safety and economic perspectives, wildlife management and conservation concerns, as well as challenges for effective reporting, registration and mitigation. In the focus of this work were often ungulates, as they combine wildlife management, safety, and economic issues (see also Bissonette and Rosa 2012). For example, Niemi et al. (2015), studied road mortality in ungulates based on collision and snow tracking data collected by Finnish hunters. They conclude that road mortality is very different among the species concerned and indeed rather high (6.5% of the wintering population of the most frequently killed deer species). Although road mortality unlikely creates a risk for viability of these ungulate populations, it may still require adaptation in wildlife management practices and hunting quotas to maintain abundant and stable populations.

A critical issue in wildlife accident statistics is the sometimes rather poor quality of data obtained from hunters, insurance companies or police. New technical development, however, allows for the involvement of first-hand reports from drivers (see also Olson et al. 2014). A growing number of countries is establishing reporting systems for citizen observations of road killed animals. Examples are presented from Belgium by Vercayie and Herremans (2015), and from California and Maine, USA, by Shilling and Waetjen (2015). Citizen science can be supported by new technologies, and provide road kill data with better extent in time and space than regular (often short-termed) scientific study, and with different extent and taxonomic accuracy (especially in smaller animals) than data collected by road maintenance or other officials. Citizen-reported data is useful both to prioritize sites for mitigation action and to raise public awareness on accident risks and conservation concerns. However, as citizen-reported data has a different focus and produces a different picture on traffic mortality in wildlife than official reports, it should be used as a complement to rather than as a replacement.

Another promising technical development are automated animal detection systems that intend to warn vehicle drivers when animals approach the road (see also Huijser et al. 2003). Grace et al. (2015) present tests from a driving simulator that indicated that the collision risk with large animals can be reduced significantly by such systems, especially if the alerts are picture-based rather than word-based. In fact, animal detection systems may be a cost-effective way to improve traffic safety for humans and wildlife, without creating the strong barrier effect on wildlife typically imposed by traditional exclusion fences.

Physical crossing structures may however be needed at certain location, to separate wildlife from traffic in a permanent manner and allow for safe passages for both animals and humans (see Beckmann et al. 2010). Such crossing structures vary in type and size, depending on the target species (Iuell et al. 2003). For arboreal animals, that often experience a strong barrier in wide and busy roads, treetop bridges have proven effective. Here, Yokoshi and Bencini (2015) tell a success story of a rope-bridge that crosses a major road in Australia, effectively connecting small but important habitat patches for critically endangered possums.

As technical solutions to overcome the negative impacts of transport infrastructure and to maintain ecological connectivity typically have a rather local effect, it is important to plan them in a concerted action and in context of the surrounding landscape. Several countries have therefore developed comprehensive defragmentation plans (e.g., Voelk et al. 2001, Trocmé 2005, Bekker et al. 2011, BMU 2012). Also Favilli et al. (2015) highlight the importance of mitigation planning at a larger geographic and multidisciplinary scale. They describe how the most important barriers for large wildlife in the Carpathian mountain range were identified in trans-boundary cooperation. They further demonstrate the necessity of a broader, multiple-actors approach, as barriers for large wildlife involves also other factors than transport infrastructures alone.

Similarily, Persson et al. (2015) address the importance of a proper planning system with a broader scope for successful environmental conservation. In Sweden, compensation of negative environmental impacts of roads and railroads are rarely conducted despite both national and EU legislation calling for such action. The authors suggest stricter policies and better incentives for voluntary compensation to overcome this shortcoming.

The two final contributions to this thematic issue from the IENE 2014 conference take a contrary perspective to the previous by focusing on the positive potentials for nature conservation that are provided by habitats in infrastructure corridors. If managed appropriately, such habitats can sustain a variety of plant and animal life, including several endangered species, that may otherwise not be able to survive in the surrounding landscape (Vermeulen 1994, Bellamy et al. 2000, Milton et al. 2015). Spooner (2015) reviews the importance of roadsides for biodiversity and for producing ecosystem services in anthropogenic landscapes, using minor road networks in rural Australia as an example. While roadside management has its challenges, such as the risks of spreading of invasive species and creating ecological traps, and also is constrained by transport needs and safety concern, roadsides can be vital in providing connectivity and functioning ecosystems. Helldin et al. (2015) give examples of the importance of road and railroad verges as habitat refuges for rare or declining species in Scandinavia. They suggest that road and railroad managers adopt species for which they take a certain conservation responsibility, and use the occurrence of such responsibility species to set priorities for adapted verge management.

As the contributions to this thematic issue as well as other conference contributions show, a "greener transport infrastructure" can be achieved by effective mitigation of adverse effects and wise use of habitats managed within transportation corridors and facilities. However, it is also evident that not all negative impacts can be mitigated or compensated for. There will always be a residual and detrimental effect on nature. It is therefore only logical to conclude that in certain areas, where these residual effects are not acceptable, construction of transport facilities should be entirely avoided. Such areas need to remain (or become again) roadless to provide sufficient undisturbed space for nature conservation (DeVelice and Martin 2001, Crist et al. 2005). Even within Europe, where only small and few roadless areas reside (Selva et al. 2011), this need is increasingly recognised. To support this development and highlight the value of roadless areas as complements to current biodiversity conservation in Europe, the participants at the IENE 2014 International Conference unequivocally asked for a pan-European strategy on roadless areas (IENE 2015).

To conclude, the IENE 2014 International Conference has highlighted the ecological and social benefits of roadless areas, outlined solutions for how transportation infrastructure can be developed without compromising these benefits. The conference and has also pointed out that the transport sector is able and willing to implement these solutions for a greener transport infrastructure.

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RESEARCH ARTICLE

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Traffic mortality of four ungulate species in southern Finland

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Abstract

Ungulate–vehicle collisions are intensively studied in many countries. However, limited knowledge exists on how many animals struck actually die due to collisions and whether differences in traffic mortality occur between species living in the same area. In this study, we estimated a kill rate (the proportion of individuals killed/struck) and, in relation to their winter population sizes, the collision and traffic mortality rates for four ungulate species (moose *Alces alces*, white-tailed deer *Odocoileus virginianus*, roe deer *Capreolus capreolus*, and fallow deer *Dama dama*). We used an unofficial collision register collected between 2001 and 2012 (a total of 12 years) by voluntary hunters from the Hyvinkää Game Management Area (323 km^2) located in southern Finland. The population estimates used were based on annual snow track censuses. A total of 497 ungulates were involved in collisions during the study period. Of these, 76% were killed directly or put down afterwards. Roe deer had the highest kill rate; 95% of struck individuals died. White-tailed deer had the highest collision and traffic mortality rates (8.0% and 6.5% of the winter population, respectively), followed by moose (6.5 % and 4.5%), roe deer (3.9% and 3.7%), and fallow deer (3.2% and 2.1%). As we found the collision and traffic mortality rates to be unequal between species, we recommend separately reporting all ungulate species when compiling collision statistics. We additionally suggest that local managers should be aware of ungulate collision and traffic mortality rates in their areas and should use this knowledge when planning annual harvest.

Keywords
Deer–vehicle collision, moose–vehicle collision, population size, collision rate, traffic mortality rate, game management

Introduction

Expanding road networks and globally increasing traffic volumes have many negative effects on the environment and animals (e.g. Forman and Alexander 1998, Fahrig and Rytwinski 2009). Road-induced mortality is one of these impacts. Road kills are the single most important cause of death e.g. for Eurasian badgers (*Meles meles*) in Britain (Clarke et al. 1998) and for Florida Key deer (*Odocoileus virgianus clavium*) in Florida (Lopez et al. 2003). The European otter (*Lutra lutra*) is also an example of a species suffering from high traffic mortality (Philcox et al. 1999, Hauer et al. 2002). Traffic can also be a significant cause of death in many common and abundant species, e.g. many ungulates, without directly threatening their population persistence (Seiler and Helldin 2006).

Ungulate–vehicle collisions (UVCs) are a notable and increasing traffic safety problem in Europe, North America, and Japan, and are therefore intensively studied in many countries (Groot Bruinderink and Hazebroek 1996, Romin and Bissonette 1998, Seiler 2004, Huijser et al. 2009, Morelle et al. 2013). Nevertheless, human injuries and fatalities (e.g. Joyce and Mahoney 2001) or the economic consequences caused by UVCs (e.g. Bissonette et al. 2008) are not the only aspects researchers have considered.

Several studies have focused on the temporal and/or spatial patterns of UVCs (Finder et al. 1999, Haikonen and Summala 2001, Danks and Porter 2010, Rolandsen et al. 2011, Niemi et al. 2013a, Rea et al. 2014, Steiner et al. 2014) and developed models for the purpose of predicting collision sites (Seiler 2005, Found and Boyce 2011). Different mitigation measures such as overpasses (Olsson et al. 2008), fencing (Clevenger et al. 2001), or warning signs (Krisp and Durot 2007) have furthermore been developed with the aim of reducing the number or severity of accidents.

Ungulate traffic mortality in relation to their population sizes is known in many countries. For example, the traffic mortality of roe deer (*Capreolus capreolus*) has varied between 13% and 16%, depending on the country (Groot Bruinderink and Hazebroek 1996, Seiler et al. 2004, Pokorny 2006). Joyce and Mahoney (2001) calculated the same ratio for moose (*Alces alces*) in Newfoundland, Canada, and found that the traffic mortality was only approximately 3% of the annual allowable harvest quota or 0.6% of the total population. In Sweden, these numbers were 10.1% and 4.0%, respectively (Seiler et al. 2004).

Although the relative importance of road kills seems low in many ungulate populations, regional variation could be notable. For Finland, Groot Bruinderink and Hazebroek (1996) reported the annual traffic mortality to be 1.2% of the total moose population, but pointed out that the portion could be 10% in the southern part of the country with denser human populations and higher traffic volumes.

Finland has three abundant ungulate game species, whose populations are mainly regulated by hunting: the moose, which is spread across the country; the introduced white-tailed deer (*Odocoileus virginianus*), which has a dense population concentrated in southwestern Finland and the roe deer, which lives in the southern half of the country (second largest distribution) at low densities (see Pulliainen 1980, Lavsund et al. 2003, Kekkonen et al. 2012, Finnish Wildlife Agency and Finnish Game and Fisheries Research Institute 2014). In addition, two small native populations of wild forest reindeer (*Rangifer tarandus fennicus*) live in central and northeastern Finland and a few small, introduced local populations of fallow deer (*Dama dama*) in southern Finland. The hunting of all these species, except for the roe deer, is based on licenses granted by wildlife authorities (Hunting Act 615/1993, Hunting Decree 666/1993).

During the 21st century, the annual country-level harvest for moose, white-tailed deer, and roe deer has varied between 38–76 000, 14–26 000, and 1–4 000 individuals, respectively. Other species are hunted only marginally; the annual hunting bag has varied between 20–130 individuals for wild forest reindeer and 50–200 individuals for fallow deer (Finnish Wildlife Agency and Finnish Game and Fisheries Research Institute 2014). At the same time, the total amount of annual collisions has varied between 1200–3000 for moose and 2600–4300 for the deer species (Finnish Traffic Agency 2014). Traffic is probably a significant cause of mortality for Finnish ungulate species, thus playing an important role when planning the annual harvest. This is particularly true for white-tailed deer, roe deer, and fallow deer, whose distributions are located near the most densely populated human settlements in southern Finland.

Study aim

The aim of our study was to estimate the traffic mortality of four ungulate species living in the same area and to discover possible inter-species differences. First, we were interested in how many percent of animals struck actually die due to collisions (later referred to as kill rate). Second, we wanted to investigate how many collisions have occurred in relation to species population sizes (later referred to as collision rate). Finally, we wanted to explore how large a proportion of the populations have died in the collisions (later referred to as traffic mortality rate). We tested the hypotheses that 1) the collision rate is equal for all species and 2) the traffic mortality rate is equal for all species.

Materials and methods

Study area

The study was conducted in the area of the Hyvinkää Game Management Association (later referred to as Hyvinkää GMA) (Fig. 1), which covers the Hyvinkää municipality

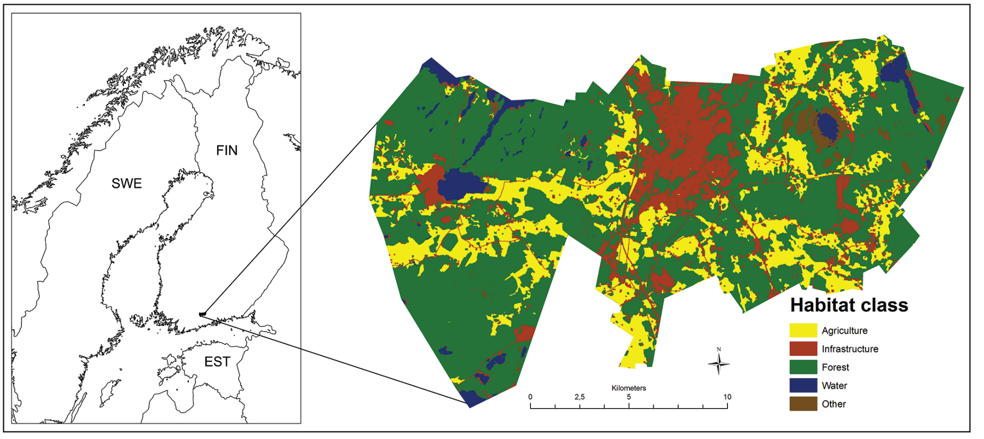

Figure 1. The map of our study area. The land use map is simplified from CORINE Land Cover 2006 data (Finnish Environment Institute 2009, CLC2006). Country borders: Eurostat.

in southern Finland, in the Uusimaa region. Hyvinkää (land area 323 km^2) is located in a densely populated part of the country with approximately 46 000 inhabitants in 2012 (Statistics Finland 2014). Most of the landscape is highly dominated by humans; the city of Hyvinkää covers the central part of the area and several smaller villages exist especially in the south. Landscape structures outside these population centers range from a mosaic of cultivated areas and settlements to more forested areas found mainly in the western parts of the municipality.

The area is divided by fenced National Highway 3 (depending on the road section, the traffic volume was approximately 20–30 000 vehicles/day in 2010; Finnish Transport Agency statistics; heavy traffic included), route 130 (running parallel to Highway 3; 3200–3500 vehicles/day) and a railway. These all run south to north, while in the east-west direction the area is limited by Highway number 25 (5–10 000 vehicles/day), which runs through the southern part of the area. Public road density is approximately 0.7 km/km^2, with an annual traffic flow approximately 330 millions of kilometers in 2010 (Finnish Transport Agency statistics). There is additionally a dense network of minor roads and forestry tracks. The speed limit on the main roads varies depending on the road section and season, being 100 or 120 km/hour on Highway 3 and 60 or 80 km/hour on the other main roads.

Ungulate data

A total of four ungulate species (moose, white-tailed deer, roe deer, and fallow deer) exist in the area of Hyvinkää GMA. All are game animals, i.e. their populations are managed by hunting. The annual maximum hunting quotas for moose, white-tailed deer, and fallow deer are defined and controlled by licenses granted by the Finnish Wildlife Agency. Roe deer harvest is not regulated by the authorities, but hunters have to report their bag (Hunting Act 615/1993, Hunting Decree 666/1993). The number of hunted individuals is thus known for each species.

The population estimate used in our study was based on an annual snow tracking census (Finnish Wildlife Agency and Finnish Game and Fisheries Research Institute 2014) coordinated by the Finnish Wildlife Agency and conducted by voluntary hunters. In the Uusimaa region, where our study area is located, each hunting club tries to assess all the ungulate individuals living in their hunting area. To avoid double counting, the census is carried out everywhere during the same weekend. Where animal populations are dense or snow conditions poor, the results of that census can be complemented with supplemental information from camera traps or other sources. The trends of an annual snow census from our study area are provided in Suppl. material 1: Annual trends in population size and collisions.

UVC data collection

Moose–vehicle accidents are registered at the species-level in the Finnish collision statistics, but crashes with other wild ungulates are treated as deer–vehicle collisions regardless of the species. Because we were interested in exploring the possible inter-species differences, the existing nation-wide collision database was not usable. We instead used a local dataset collected from the Hyvinkää GMA area by voluntary hunters who work as official assistants to the police.

UVCs in Finland have to be reported to the police, but the crash sites are usually visited by the police only in cases where personal injuries or damage to the vehicle has occurred. The collision sites are instead checked by local hunters, who work as an executive assistance to the police. These volunteers visit every UVC site, put the involved animal down if needed, and transport the carcass away from the road area. The volunteers do not have any registering duties, but will sometimes collect unofficial statistics for their own interests.

For our study, we used a specific UVC dataset collected by voluntary hunters and maintained by the chief of the Hyvinkää GMA. This register contained detailed information concerning e.g. the species and post-collision condition of an animal. The register contained UVCs from between 2001 and 2012 (12 years in total).

Data analyses

From the data collected by the voluntary hunters, we calculated a kill rate, a collision rate, and a traffic mortality rate for each species. The kill rate was simply calculated from the animals struck (how many percent of animals struck died in the collision or were fatally injured and put down afterwards). The collision rate was calculated by combining the collision data and the results of an annual snow census (i.e. how many collisions occur for each 100 individuals assessed in the snow census). The traffic mortality rate was also based on the collision and snow census data (how many individuals died in collisions for each 100 individuals assessed in the snow census). We have converted our results to percentages (e.g. a calculated rate of 0.05 = 5%) to simplify the text.

We used Fisher's exact test (e.g. Ranta et al. 1999) to test possible differences between species. Contingency tables used for the analyses are presented in Suppl. material 2: Contingency tables. We used Fisher's exact test with Bonferroni corrections for *p*-values for the post hoc analyses (MacDonald and Gardner 2000). Analyses were conducted using R software, version 3.1.3 (R Development Core Team 2015).

Results

A total of 497 ungulates were involved in 493 collisions during the 12-year study period (Table 1, Suppl. material 1). One out of two collisions ($N = 245$; 50%) was a crash involving white-tailed deer, followed by moose (118; 24%), roe deer (75; 15%), and fallow deer (40; 8%). The species was unknown in 15 cases (3%).

A total of 378 individuals (76%) were killed directly in the collisions or put down afterwards (later referred to as road-killed) (Table 1). Roe deer was the most vulnerable species: 95% of individuals involved in crashes were killed and only one was found uninjured. The lowest kill rate (65%) was recorded for fallow deer, but concurrently the number of disappeared individuals was high.

In comparison to population estimates derived from the snow track census data, white-tailed deer had the highest collision rate: 8.0% (8.0 collisions/100 individuals), followed by moose (6.5%), roe deer (3.9%), and fallow deer (3.2%) (Fig. 2). A statistically significant difference was observed between species (DF = 3, $p < 0.001$). A paired post hoc comparison showed all species pairs except white-tailed deer & moose and moose & roe deer to differ (0.05 at the α-level) after the Bonferroni correction was applied (Table 2).

Table 1. Road-killed and struck but uninjured ungulates in the Hyvinkää GMA between 2001 and 2012 (a total of 12 years). Column "Condition unknown" contains animals that have disappeared from the collision site and have not been found later by tracking, and animals whose condition has not been recorded in the database used.

	Road-killed individuals	Uninjured individuals	Condition unknown	Total number of individuals struck
White-tailed deer	198 (80%)	4 (2%)	46 (19%)	248 (50% of all)
Moose	82 (69%)	12 (10%)	24 (20%)	118 (24%)
Roe deer	72 (95%)	1 (1%)	3 (4%)	76 (15%)
Fallow deer	26 (65%)	1 (3%)	13 (33%)	40 (8%)
Unknown	0 (0%)	1 (7%)	14 (93%)	15 (3%)
Total	378 (76%)	19 (4%)	100 (20%)	497 (100%)

Figure 2. Annual variation in collision and traffic mortality rates for four ungulate species in the Hyvinkää GMA between 2001 and 2012 (a total of 12 years).

Table 2. The results of the pairwise comparisons between species' collision rates in the Hyvinkää GMA between 2001 and 2012 (a total of 12 years). Comparisons were made by using Fisher's exact test and *p*-values were adjusted using the Bonferroni correction.

Species 1	Species 2	DF	*p*-value
White-tailed deer	Moose	1	0.048
White-tailed deer	Roe deer	1	<0.001***
White-tailed deer	Fallow deer	1	<0.001***
Moose	Roe deer	1	<0.001**
Moose	Fallow deer	1	<0.001***
Roe deer	Fallow deer	1	0.332

*** p < 0.001 after Bonferroni adjustment
** p < 0.01 after Bonferroni adjustment
* p ≤ 0.05 after Bonferroni adjustment

When analyzing road-killed individuals only (i.e. excluding animals that had disappeared after the collision or were found uninjured), it became apparent that white-tailed deer had the highest traffic mortality rate (6.5% or 6.5 road-killed individuals/100 individuals) followed by moose (4.5%), roe deer (3.7%), and fallow deer (2.1%) (Fig. 2). Again, a statistically significant difference was found (DF = 3, p < 0.001), and in a paired comparison all species pairs except moose & roe deer differed (0.05 at the α-level) after the Bonferroni correction was applied (Table 3).

We calculated the ratio between road-killed individuals and the annual harvest for each species. The proportion of road-killed white-tailed deer was 10.3% of the annual hunting bag. The same proportions for moose, roe deer, and fallow deer were 6.9%, 30.9%, and 49.1%, respectively.

Table 3. The results of the pairwise comparisons between species' traffic mortality rates in the Hyvinkää GMA between 2001 and 2012 (a total of 12 years). Comparisons were made using Fisher's exact test and *p*-values were adjusted using the Bonferroni correction.

Species 1	Species 2	DF	*p*-value
White-tailed deer	Moose	1	0.003*
White-tailed deer	Roe deer	1	<0.001***
White-tailed deer	Fallow deer	1	<0.001***
Moose	Roe deer	1	0.249
Moose	Fallow deer	1	<0.001**
Roe deer	Fallow deer	1	0.008*

*** $p < 0.001$ after Bonferroni adjustment
** $p < 0.01$ after Bonferroni adjustment
* $p \leq 0.05$ after Bonferroni adjustment

Discussion

Collision fatality for ungulates

UVCs, especially deer–vehicle collisions, are relatively rarely fatal for humans. The opposite is true for animals. In our data, the smallest species, roe deer, was the most vulnerable: 95% of crashes lead to the death of the animal. This percentage is almost the same (94%) as that found by Almkvist et al. (1980) in Sweden. This number was lower for other species, but the number of disappeared individuals was concurrently higher. It is not known how large a proportion of these animals has been wounded and would have died later due to the consequences of the collisions. However, when ignoring these disappeared individuals, the largest ungulate species, the moose, has the best possibility of surviving a collision: 10% of individuals struck were found to be uninjured. This is similar to findings from Sweden (Almkvist et al. 1980; 8%) and Newfoundland, Canada (Joyce and Mahoney 2001; 11%). We thus note that the number of collisions with wild ungulates is more or less the same as the amount of road-killed animals. It is hence good to keep in mind that not all accidents are reported (e.g. Almkvist et al. 1980), and therefore the real number of collisions and further, the number of road-killed animals, may be larger than the number of registered accidents.

Although the size of the struck animal seemed to be an important factor affecting its possibility of surviving a collision, it is not necessarily the only one. Vehicle speed is the most important single variable that is connected to the severity of ungulate–vehicle collisions from the human point of view (Garret and Conway 1999, Joyce and Mahoney 2001), i.e. increasing speed increases the risk of human injuries or fatalities due to collisions. It is thus logical to assume that the probability that an animal struck would die in a collision is larger on highways with high speed limits than on secondary roads. Unfortunately, the data we used contained no exact spatial information of collision sites or their speed limits so we were unable to test the possible effect of speed on the kill rate of animals.

Species-specific collision and traffic mortality rates

White-tailed deer had the highest collision and traffic mortality rates: eight out of one hundred animals (in the wintering population) were involved in collisions, and the traffic mortality rate was 6.5% of the population. Etter et al. (2002) studied the survival rate of white-tailed deer in suburban Chicago and found traffic-induced mortality to be almost twice as high as our results (a rate of 0.10 for does and 0.17 for bucks compared to 6.5% or 0.065 in our data), while Dusek et al. (1989) reported traffic-related mortality of only 2% in autumn populations along the Lower Yellowstone River. We found the collision and traffic mortality rate of moose to be 6.5% and 4.5% of the population, respectively. This was similar to what Seiler et al. (2004) estimated in Sweden, but lower collision rates have been found elsewhere (Groot Bruinderink and Hazebroek 1996, Joyce and Mahoney 2001). The roe deer traffic mortality rate found by us was in concurrence with what other European countries reported in the early 1990s, while the fallow deer mortality rate was somewhat higher than reported elsewhere (Groot Bruinderink and Hazebroek 1996).

However, a straight comparison between collision or traffic mortality rates from different areas without knowledge of other explanatory factors does not necessarily illustrate the whole picture. The actual amount of collisions, and hence the amount of road-killed animals, is affected by several factors. Population size is one of the important variables explaining the number of UVCs (Lavsund et al. 2003, Seiler 2004, Rolandsen et al. 2011). In addition, though not always simple and linear, traffic volume is probably one of the key factors affecting the number of UVCs (e.g. Seiler 2004, Seiler 2005, Balčiauskas 2009). Other factors such as wildlife fences, under- and overpasses, and the distribution of feeding sites can also have an effect on the amount of collisions especially at local scales, and further, the number of road-killed animals. Thus, it is very likely that collision and traffic mortality rates vary between different areas and/or over time, even at the same population density.

In this study, we were interested in the differences concerning collision and traffic mortality rates between species concurrently living in the same area. The traffic flow and environmental variables were thus same for all species, giving us the possibility of discussing and comparing species behavior-related factors.

We found white-tailed deer to have the highest collision and traffic mortality rates, followed by the moose. However, after calculating Bonferroni corrections for p-values, the statistical difference between species collision rates disappeared while difference between species traffic mortality rates remained. This is likely to be due to the smaller body size of white-tailed deer, and further to the larger road kill rate found in our study. Comparing this species pair in a more detailed fashion would be interesting in the future, to investigate whether their collision rates really differ. Moose are known to have large home ranges and some of the animals implement seasonal migratory behavior (Heikkinen 2000, Singh et al. 2012). This results in moose being more likely to cross several roads during their routine movements. Laurian et al. (2008) on the other hand found that moose tend to avoid road crossings although they occasionally visit the proximity of road areas. This is not necessarily true for white-tailed deer: Feldhamer et al. (1986) observed that seasonal home ranges of some white-tailed deer individuals frequently overlapped a national highway. Although all the species we studied are adjusted to living in human-dominated landscapes, it is possible that the white-tailed deer utilizes more human-dominated areas of the landscape than the moose and is therefore more likely to cross roads during its daily routines.

In our study, roe deer and fallow deer had the lowest collision and traffic mortality rates. This could be connected with the movement behavior of these animals. Studies conducted in southern Finland found the monthly home ranges to be smaller and the daily movement distances shorter for roe deer compared to white-tailed deer (Saari 2011, Honzová 2013). In addition, apart for the home range size, landscape use could be an important factor affecting collision probability. Putman (1997) reviewed the studies concerning the daily movements of different deer species and found that although all the species regularly crossed minor roads, major roads or railways could act as a home range border, at least for the roe deer and fallow deer. The movement behavior of fallow deer in Finland has not been studied, but in general it seems that the species is relatively local and therefore might be less vulnerable to traffic than other ungulates. Groot Bruinderink and Hazebroek (1996) noted that the annual traffic mortality of an Irish fallow deer population living in a park area surrounded by a heavy traffic load was only approximately 7% of the population. On the other hand, the majority of fallow deer living in our study area were concentrated in the parts with no heavy traffic or high speed limit roads, so it is very possible that our findings could be partly explained by the animals' distribution in the field.

Ungulate–vehicle collisions cannot happen without an animal being on the road, but the temporal peak of the road crossing rate of animals and the timing of collisions are not necessarily the same. Neumann et al. (2012) combined spatiotemporal moose movement data with the Swedish collision register and found that the road-crossing probability was highest in early summer and mid-winter, while moose–vehicle collisions peaked in autumn and winter or during annual migration. They concluded that a high collision risk was related not only to animal movements, but also to light and road surface conditions. Moose–vehicle collisions in Finland are more likely to occur in autumn (Haikonen and Summala 2001), when the driving conditions are typically poor. Collisions with white-tailed deer also mainly occur during rutting season in late autumn. Contrastingly, both male roe deer movements and the roe deer collisions peak in late spring or early summer (Niemi et al. 2013b) during good light and road conditions. It may thus be possible that drivers are capable of avoiding some potential roe deer collisions because of good driving circumstances, leading overall to smaller collision and traffic mortality rates than in the case of moose and white-tailed deer.

Road kills and an annual harvest

We found notable inter-species differences when comparing the number of road-killed ungulates in relation to the annual hunting bag. The proportions of road-killed animals for the relatively abundant white-tailed deer and moose were 10.3% and 6.9% of the annual harvest, respectively. For moose, this was comparable with the ratio found in Sweden (Seiler et al. 2004). However, it is very likely that these numbers varied between areas, which should be considered by the local game authorities when planning annual harvests.

For fallow deer, the number of road-killed individuals was almost as high as the annual harvest. Although the data size was small and strong conclusions should therefore be avoided, our observation implicates the importance of taking species-specific traffic mortality into account when planning harvest quotas. Because the Finnish nationwide collision register does not differentiate between deer species, local-scale managers could benefit from their own, unofficial collision statistics.

Contrastingly to the relatively low traffic mortality rate (3.7%) of the roe deer, the species' traffic mortality in relation to the hunting bag (30.9%) was high. Roe deer hunting in Finland is free of licenses, leaving more management responsibility to local hunting clubs and even individual hunters. The past decade has been somewhat difficult for roe deer in southern Finland; the increasing Eurasian lynx (*Lynx lynx*) population and several severe winters have inhibited the population increase that began approximately two decades ago. It seems that hunters have tried to react to the changing situation by reducing their game bag; the annual amount of hunted roe deer compared to the estimated population size has decreased during the last few years in our study area (Finnish Wildlife Agency and Finnish Game and Fisheries Research Institute 2014). On the other hand, the annual amount of roe deer collisions has concurrently slightly increased (Suppl. material 1).

Conclusions

In our paper, we compared the collision statistics of four ungulate species (moose, white-tailed deer, roe deer, and fallow deer) living in the same area. Our main finding shows that both the collision (collisions in relation to population size) and traffic mortality rates (animals killed in collisions in relation to population size) of these four ungulate species differed. White-tailed deer and moose suffered the highest collision and traffic mortality rates. These rates were relatively low for roe deer and especially for fallow deer, although no strong conclusions could be drawn because of the limited amount of data especially in the case of fallow deer.

We were only able to show that the species-specific collision and mortality rates differed, but were unable to evaluate the actual reasons behind our findings. Additional work is thus needed to investigate, which factors affect the amount of collisions, and further, how traffic mortality affects ungulate populations.

However, we believe that managers responsible for defining the hunting quotas could use our results as a tool when planning the management of different ungulate species. We additionally wished to point out that combining several species under the same category in collision statistics may lead to loss of information and should therefore be avoided. Thus, in cases where the official collision register does not contain species-specific information or does not exist at all, local managers may benefit from a detailed collision registering system such as the one used in our study area.

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Supplementary material 1

Annual trends in population size and collisions
Authors: Milla Niemi, Juho Matala, Markus Melin, Visa Eronen, Hannu Järvenpää
Data type: species data
Explanation note: Annual trends in population size, harvest, and collisions.

Supplementary material 2

Contingency tables used in the analysis of collision and traffic mortality rates
Authors: Milla Niemi, Juho Matala, Markus Melin, Visa Eronen, Hannu Järvenpää
Data type: species data
Explanation note: Contingency tables.

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Citizen science and smartphones take roadkill monitoring to the next level

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Abstract

Road networks, even in industrialized countries, become denser year after year and traffic volumes continue to increase at a steady pace. It is imperative that we monitor the impact of this trend on wildlife, but monitoring roads for flattened fauna is a time consuming effort and roadkill monitoring projects conducted up till now have been relatively small scale both in terms of time and space. This hampers the progress of road ecology analyses at the population level and at larger landscape extents.

We demonstrate that citizen science projects in combination with smartphones and other new technologies allow analysis at this level and extent, and simultaneously offer more complete data for safer transportation and mitigation of roadkill hotspots. Monitoring roadkill with citizen scientists poses certain challenges regarding data quality and people management, but we show that these challenges can be addressed, which allows researchers to benefit from the many other advantages and possible applications of monitoring roadkill with citizen scientists, including raising public awareness on the matter.

Keywords

Citizen science, roadkill, monitoring, hotspots, road ecology

Introduction

Despite the already high density of roads in industrialized countries, road length and density still increase year after year. Moreover, traffic volume increases as well. E.g. in Europe the length of the motorway network increased with 68% between 1990 and

2010, the number of passenger cars per thousand inhabitants increased with 39% between 1990 and 2010 and the number of kilometers driven by passenger cars increased with 22% between 1995 and 2010 (European Commission 2014). As motorways get more congested, traffic increases on minor roads (van Langevelde and Jaarsma 2009, van Langevelde et al. 2009). The environmental effects are numerous and can be categorized in many ways; e.g. biotic and abiotic effects, direct and indirect, short term and long term. Examples of environmental effects are habitat loss, landscape fragmentation, loss of connectivity, pollution and many wildlife road casualties, also known as roadkill (Forman and Alexander 1998, Spellerberg 1998, Seiler 2001, Coffin 2007). Animals killed are not just the young, weak, or ill individuals; healthy adults are killed equally often by traffic in contrast to natural predation (Bujoczek et al. 2011). Many studies have been undertaken to investigate the number of animals killed on specific road sections, but few authors have assessed the impact of road-killed fauna at larger scale extents or at the population level (van der Ree et al. 2011). In an editorial for a special issue of Ecology and Society, van der Ree and colleagues (2011) stated: 'Counting roadkills by itself is not enough to assess whether roads and vehicles endanger the existence of populations or species. Or counting the number of animals crossing underpasses is not enough to assess if populations on both sides have become more viable. (…) The next step is to evaluate how the density and configuration of entire road networks affect the functional relationships within and among ecosystems at the landscape scale.'

Roadkill monitoring projects conducted up till now are relatively small scale both in terms of time and space, hampering analysis at larger scales. Monitoring roads for faunal mortality is a time consuming effort, especially because many more roadkills are discovered when surveys are done on foot or at very slow speeds compared with surveys done by car at higher speeds (> 20 $km.h^{-1}$) (Slater 2002). Therefore researchers have to weigh search effort (in time and space) and the resulting amount of data gathered against available budgets. Here we present a way to gather large amounts of data on a large scale, with relatively little effort. As Devictor and colleagues (2010) already pointed out, citizen science projects are the way to go 'beyond scarcity' of means and data. On the basis of a project carried out in Flanders (Belgium), we demonstrate that a citizen science project based on so called roving records and transect data in combination with new tools like smartphones can take roadkill monitoring to the next level.

Methods

Citizen science and its application to roadkill monitoring

A citizen scientist is a volunteer who collects and/or processes data as part of a scientific enquiry (Silvertown 2009). Citizen science is a relatively new term for an old practice with an established tradition in astronomy, local history, archaeology, natural history or ecology and especially in ornithology (Greenwood 2007). New tools like internet, smartphones and open source tools as Google Maps, gave this practice a new boost

(Silvertown 2009, Bonney et al. 2009). The potential of citizen science in ecological studies is enormous with applications ranging from the traditional ornithological projects for gathering information on species distributions and phenology to detection of invasive species and even prediction of spread of introduced species (Silvertown 2009, Sullivan et al. 2009, Gallo and Waitt 2011, Ashcroft et al. 2011). With help of citizen scientists we can monitor processes on broad geographic scales and on private grounds which would be difficult to monitor in a traditional way (Dickinson et al. 2010).

Despite its advantages for gathering large amounts of data, only in recent years the number of roadkill monitoring projects based on citizen science and web-based reporting increased rapidly (van der Ree et al. 2015) and parallel with the rise of smartphones a virtual explosion of the number of smartphone apps for roadkill registration was observed (Bissonette 2014). How smartphones can improve the quality of data collection in roadkill monitoring by professionals has already been demonstrated by Olson and collegues (2014). Examples of very successful citizen science roadkill monitoring projects are a project initiated in Flanders (Belgium) in 2008 and a similar project initiated in 2010 in California (UCDAVIS) which was later also implemented in Maine (Maine Audubon). Both projects initially only used so called roving records.

Roving records versus standardized data

A distinction can be made between citizen science projects gathering data in a standardized way (like breeding bird surveys) and projects in which data are gathered incidentally or opportunistically as “roving records”. In the latter, search effort is not directed, nor accounted for and only ‘presence data’ is gathered. Websites like www.ornitho.de, www.artportalen.se or www.observation.org, offer the opportunity to enter incidental observations and consult the data entered by other users. The advantage of this way of gathering nature observations is that it results in huge amounts of data. For example the website waarnemingen.be (local Belgian version of observation.org) gathered more than 15 million observations in Belgium (30.528 km²) since its start in 2008 (waarnemingen.be 2014). Until now little research has been conducted on the information gathered by this kind of projects, but some very promising studies have been published in recent years (Snäll et al. 2011, Sardà-Palomera et al. 2012). The pitfalls of using roving records are numerous and concern, among others, problems related to search effort (in population trend analyses), detectability and observer expertise. However, many of these problems can be addressed during the analyses if good knowledge of the limitations of the data are available or if the roving records are combined with other data sources (Kéry et al. 2010, Yu et al. 2010, Snäll et al. 2011, Sardà-Palomera et al. 2012). The growing number of publications based on roving records shows that the careful use of this kind of data is now becoming accepted in the scientific world (Silvertown 2009).

The main disadvantage of roving records compared to repeated monitoring of transects is the unknown search effort in the former approach. Therefore roving records cannot be used for impact studies where extrapolations to absolute number of victims per km per year are needed. A possible bias towards higher reporting rates for less common species cannot be detected either. Road type effects and the effect of a higher search effort (more observers on busier roads) are difficult to disentangle. Therefore in the Belgian citizen science project on roadkills (as well as in the project in Maine) an extra module was added to the website to gather standardized transect data. Monitoring fixed transects, however, is more demanding for volunteers and less people will be willing to participate (Bonney et al. 2009). Both methods have their advantages and disadvantages, but we will show that combining these methods for roadkill monitoring and carrying them out with citizen scientists has great advantages.

Challenges associated with citizen science programs

Below we will elaborate on the advantages of roadkill monitoring with citizen scientists, but in order for citizen science programs to be successful they have to meet a few challenges specific to citizen science programs. The most important challenges involve people management and ensuring data quality (Bisonette 2014, Conrad and Hilchey 2011, Dickinson et al. 2010, Gura 2013). Applied to roadkill monitoring with citizen scientists the major challenges concerning data quality are correct species identification, precise spatial location and double counting. Another challenge is spatial bias, but how important these challenges are depends on the research question. As mentioned earlier these problems can be addressed a priori or a posteriori if good knowledge of the limitations of the data is available.

People management, or more specifically recruiting volunteers and keeping them engaged, is probably the biggest challenge for citizen science projects (Conrad and Hilchey 2011, Gura 2013). But if researchers are aware of these challenges, they can be overcome and turn citizen science projects in data goldmines. Below we will show how these challenges were addressed in the Belgian citizen science roadkill monitoring.

Specifications of the Belgian citizen science roadkill monitoring project

The project 'Dieren onder de wielen' (freely translated as 'Flanders flattened fauna') was a public-private initiative of the Flemish government and the NGOs Natuurpunt and Vogelbescherming Vlaanderen. It started in 2008 and data is still being gathered. The objectives of this project were (1) to identify roadkill hotspots, (2) to collect data to measure the impact of roads and traffic on fauna and (3) to raise public awareness for the effects of habitat fragmentation by roads.

The project focused on Flanders, the northern part of Belgium, with a surface area of 13.521 km². The road network in Flanders is the most dense in Europe with 5,08 km per km², except for the microstate Malta (European Road Federation 2011). With 359 inhabitants per km², Belgium also has the third highest human population density of Europe, after Malta (1.316) and The Netherlands (493) (EUROSTAT).

The advantage of this high population density is the high potential for citizen science projects, which might result in high data density, suitable for roadkill hotspot analysis.

Natuurpunt (Flanders, Belgium) and Stichting Natuurinformatie (The Netherlands) manage the website www.waarnemingen.be (the local version of www.observation.org) through which opportunistic observations of fauna and flora can be registered in a database. When adding an observation of an animal, the observer is able to choose the option 'roadkill' from different 'behaviours'. The project focused on records of roadkilled vertebrates.

There are three different apps available which allow to upload nature observations to the website mentioned above, one app for each of the three main operating systems of smartphones (ObsMapp for Android, iObs for iPhone and WinObs for Windows Phone). These apps were developed by volunteers in collaboration with Stichting Natuurinformatie. Observations can be recorded in the field (without internet connection) and uploaded to the website with just one tap on an icon when an internet connection is available. The advantages of entering data with smartphones is that it is fast, easy and precise. The observer doesn't need to copy observations from his field notebook to the website. Pictures made with the smartphone can easily be added to the observation and all observations entered on a smartphone are directly linked to the current date and GPS location. Since ObsMapp version 5.0 this app is equipped with a speech input option which makes safe recording of roadkills possible while driving.

Since October 2013 an extra module was added to the website to enter systematic roadkill transect data (accessible through www.dierenonderdewielen.be). Volunteers were asked to choose a route (e.g., their route from home to work), enter it on the website and check this transect at least once every two weeks for roadkills but no more than once a day. Double counts of the same individual roadkill by the same volunteer will be rare for most (small) species when volunteers check their transect maximum once a day, because most carcasses persist for less than one day (Santos et al. 2011). For species weighing less than 200 g, carcass persistence is even less than 2,5 hours (Slater 2002). This transect data will be analyzed with a method similar to the study of Roger et al. (2012). The volunteers were asked to fill out the monitoring form on the website for every survey, whether they found roadkills or not, to avoid a bias in the number of roadkills found per km. Based on the resulting data we have no reason to assume volunteers didn't comply with this request. For safety reasons, volunteers were not obliged to stop for identification of the roadkill. This could have an influence on the correct identification of the species, but volunteers were not obliged to identify the roadkill to species level either. It is possible to add observations like 'mammal species' or 'bird species' and volunteers were asked to identify the roadkill as accurate as possible.

Promotion for the project was made to the vast network of members and volunteers of Natuurpunt (95.000 family memberships) and Vogelbescherming Vlaanderen and through regular coverage in different media (journals, magazines, newsletters, television, radio, …); this was instrumental in generating public awareness for the issue and recruit volunteers to enter observations of roadkills. Efforts were made to keep volunteers monitoring transects engaged by sending them a monthly newsletter with information on what to expect, feedback on results and applications of the data.

To assure data quality observations of rare species, species which are easily confused with other species and observations with pictures are verified by administrators (volunteers with expert knowledge). Of all roadkill observations that were verified (anno March 2015) only 2,04% got a different species name, but this could be as well in the positive sense (from higher taxon to lower, e.g. 'mammal unknown' to 'rabbit') as the negative sense (from one taxon or species to another or higher taxon). Another 0,85% were labeled 'not assessable'.

Performance of citizen science roadkill monitoring

During the project period (2008/05/15–2014/11/30) 48.517 roadkills of vertebrates on Flemish roads were registered on the website by more than 2000 volunteers. This is a mean density of 3,6 roadkills per square kilometer or 0,7 roadkills per kilometer of road. Therefore, the project resulted in one of the largest and probably the densest dataset on roadkills in the world. This allows for instance to make detailed hotspot analyses, but when analyzing the data researchers should keep the limitations of the data in mind. Based on a subsample of the data we estimate the double counts at about 4% of the total dataset. When analyzing a certain hotspot these double counts can be disentangled by looking at the date and observer for each roadkill. This type of data is not used to estimate the actual number of roadkills per kilometer for a given time period, because these are roving records and therefore there is no account of search effort.

Thirteen months after the start of the systematic monitoring of routes 78 volunteers were monitoring 110 routes for a total of 941 km. Already 2.370 route counts have been registered and 18.995 km surveyed, of which 11.833 km by car, 6.941 km by bike and 221 km on foot. These volunteers found 1.403 roadkills. If we assume a mean speed of 70 km/h for surveys by car, 20 km/h for bikes and 5 km/h for surveys on foot, the effort of these volunteers is comparable to three and a half month of full time work, assuming one can stay focused on roadkills for 7,5 hours per day and 5 days a week.

As expected, it was harder to find volunteers for the systematic monitoring and they recorded only 1.403 roadkills in 13 months, compared to 13.809 'roving' roadkill records of which about 4% were double counts. This comparison clearly shows the advantage of citizen scientists gathering incidental observations of roadkills: almost ten times as much records are gathered with far less effort. Nevertheless systematically gathered roadkill data is necessary for certain analyses as mentioned before and can be carried out with citizen scientists.

The data gathered in the systematic roadkill monitoring is also spatially explicit: roadkill and route positions are registered on the website and can be extracted as digital maps for analysis in GIS software. Therefore all kinds of analysis are possible as for instance relations between number of roadkills and road type, maximum speed or land use.

The roving records data have already been applied to identify roadkill hotspots for single species (e.g., squirrels) or different species together (e.g., amphibians) and mitigation measures were taken for several of them. The project gave information on landscape connectivity issues, distribution of seldom observed or rare species and revealed the most vulnerable species (in terms of most recorded) and seasonal patterns in numbers of roadkill per species. The standardized way of monitoring is currently also being applied to monitor the success of mitigation measures.

Applications and advantages

The applications and advantages of the roadkill monitoring data gathered by citizen scientists are infinite. We list a few important ones below for roving records and transect data separately.

Roving records data

- A ranked list of most vulnerable species can be drawn from both, classical monitoring of transects by researchers and citizen science projects based on roving records, but the latter offer the opportunity to continue the monitoring with relatively few means for many years and monitor the changes in the ranking. This can be a crude way of monitoring species abundance. For instance, in the present study, red fox (*Vulpes vulpes*) was the 3th most frequently reported victim and stone marten (*Martes foina*) was the 9th, while both were not even in the top ten of roadkills during a study in 1995 (Rodts et al. 1998); this increase in road victims parallels their return to many parts of Flanders (Van Den Berge, personal communication 2009).
- Seasonal patterns in numbers of roadkill per species arise from these data.
- Several authors highlight the importance of roving records as a complementary data source for monitoring the distribution of rare species (Snäll et al. 2011, Sardà-Palomera et al. 2012). The large quantity of data gathered in citizen science projects results in a higher chance to detect rare species. This way "the dead can also be used to monitor the living" or more precisely to improve distribution maps (George et al. 2011). Even new species can be discovered by monitoring road kills (Auliya 2002, Covaciu-Marcov et al. 2012).
- By running this kind of monitoring for years, roadkill hotspots and landscape connectivity issues can be detected not just along a few trajectories monitored by researchers, but along most roads in a large project area.
- If the data is made available, governments or other road managers will be able to take evidence based mitigation actions (Greenwood 2007).
- In recent studies (Snäll et al. 2011, Sardà-Palomera et al. 2012) roving records were used to monitor and model species distributions. A next step might be to model and predict roadkill hotspots for certain species on the basis of roadkill monitoring data in combination with Species Distribution Models (SDM).

- A database on roadkills can lead to safer transportation, for instance when it is used as basis for a warning system for wildlife-vehicle collision hotspots on GPS-systems for cars. Such a system could be based on actual roadkill observations or on models. In Sweden such an app for smartphones was launched in 2012 based on the police database of wildlife vehicle collisions (Nationella Viltolycksrådet). A similar app, but based on driver sightings, called AvoiDeer, was developed in Norway (www.avoideer.com).
- Last but not least, involving citizens in roadkill monitoring has the great advantage that it raises awareness for the issue, or as Devictor and colleagues (2010) put it "citizen science promotes the reconnection between people and nature, and between people and science". When the data is made available as a digital map on a website, a direct cooperation and interaction can be established between volunteers gathering the data, researchers analyzing the data and local governments or other road managers responsible for mitigation actions.

Transect data

- Continues monitoring of stretches of road for roadkills by volunteers is a good way to monitor the success of mitigation measures.
- In contrast to roving records, transect data offers the possibility to disentangle the effect of road type or maximum speed on a road and the number of passing observers.
- Potential biases in reporting rates for different species based on roving records can be detected and corrected.
- On the basis of transect data (with known search effort and presence and absence data) an extrapolation can be made to an absolute number of roadkills per km per year and therefore the impact of roads on mortality rates can be assessed. Roger et al. (2012) demonstrated that by combining roving records, SDM and monitoring of fixed transects for roadkills, a (rough) estimate can be made of the absolute impact of roads and traffic on species, taking roadkill studies to the next level.

The advantage of citizen science programs and opportunistic data as a complement to standardized protocols was established earlier by Snäll et al. (2011) for bird monitoring, but from the previous it is clear that it has many advantages for roadkill monitoring too.

Advantages of one portal for all nature observations

Worldwide several portals exist which are restricted to bird records, with or without the option for entering roving records or for monitoring roadkills. We argue that a combination of the possibility to enter observations of all kinds of species, both by checklist or opportunistic observations and the option to label the observation as 'roadkill' has many advantages. Observers add their different kinds of data via the same website which offers a lot of feedback possibilities and therefore stimuli for the observer, such as for instance an overview of all the observer's personal entries, different statistics and maps and comparisons with other observers. Records of observations of dead or living animals can be combined instantly to produce distribution maps for each species. The combination of records of living and dead animals offers the opportunity to analyze in which areas the species is present but doesn't get killed on roads and areas where it is present and is frequently run over. Therefore, a website and database like www.observation.org, which combines the possibility of entering roving records data of living or dead species for all taxa as well as checklists or transect data, offers the ideal tool for roadkill monitoring. The tools (website, app) of this roadkill monitoring project are available and can be deployed anywhere in the world.

Conclusion

Van der Ree et al. (2011) argued that analysis on larger landscape extents and population level could bring road ecology to the next level. Large datasets spanning large areas based on roving records might be the gateway to this next level. Based on the results of the ongoing roadkill monitoring project in Belgium we conclude that gathering roving records of roadkills and systematic monitoring of roadkills with citizen scientists, facilitated by tools like smartphones indeed deliver big data and take roadkill monitoring to the next level.

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Wildlife-vehicle collision hotspots at US highway extents: scale and data source effects

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Abstract

Highways provide commuter traffic and goods movement among regions and cities through wild, protected areas. Wildlife-vehicle collisions (WVC) can occur frequently when wildlife are present, impacting drivers and animals. Because collisions are often avoidable with constructed mitigation and reduced speeds, transportation agencies often want to know where they can act most effectively and what kinds of mitigation are cost-effective. For this study, WVC occurrences were obtained from two sources: 1) highway agencies that monitor carcass retrieval and disposal by agency maintenance staff and 2) opportunistic observations of carcasses by participants in two statewide systems, the California Roadkill Observation System (CROS; http://wildlifecrossing.net/california) and the Maine Audubon Wildlife Road Watch (MAWRW; http://wildlifecrossing.net/maine). Between September, 2009 and December 31, 2014, >33,700 independent observations of >450 vertebrate species had been recorded in these online, form-based informatics systems by >1,300 observers. We asked whether or not WVC observations collected by these extensive, volunteer-science networks could be used to inform transportation-mitigation planning. Cluster analyses of volunteer-observed WVC were performed using spatial autocorrelation tests for parts or all of 34 state highways and interstates. Statistically-significant WVC hotspots were modeled using the Getis-Ord Gi* statistic. High density locations of WVC, that were not necessarily hotspots, were also visualized. Statistically-significant hotspots were identified along ~7,900 km of highways. These hotspots are shown to vary in position from year to year. For highways with frequent deer-vehicle collisions, annual costs from collisions ranged from US$0 to >US$30,000/km. Carcass clusters from volunteer data had very little or no overlap with similar findings from agency-collected WVC data, during a different time-range. We show that both state agency-collected and volunteer-collection of WVC observations could be useful in prioritizing mitigation action at US state-scales by state transportation agencies to protect biodiversity and driver safety. Because of the spatial extent and taxonomic accuracy at which volunteer observations can be collected, these may be the most important source of data for transportation agencies to protect drivers and wildlife.

Keywords

Transportation, Wildlife-Vehicle Collisions, Roadkill, Informatics, Citizen Science, Wildlife Observation, Wildlife Movement

Introduction

Wildlife-vehicle collisions (WVC) are a large and growing concern among Departments of Transportation (DOT), conservation organizations and agencies, and the driving public (Huijser et al. 2008). WVC is a safety concern for drivers (Bissonette et al. 2008) and a conservation concern for most animal species (Fahrig and Rytwinski 2009). Recently, Loss et al. (2014) estimated that between 89 and 340 million birds may die per year in the US from collisions with vehicles. Many DOTs are trying different methods of reducing WVC, including fencing roadways and providing crossing structures across the right-of-way to allow safe animal passage. WVC occur when traffic coincides with a place where animals decide to cross the surface of a roadway. Predicting and prioritizing these places for mitigation of impacts to wildlife and drivers is an important step in reducing the conflict. To inform these types of predictions and corresponding mitigation at a large scale (e.g., a US state), it becomes necessary to collect accurate, extensive, long-term WVC data.

Monitoring biodiversity and investigating causes of changes in biodiversity allows society to make decisions about conservation (Wilson 1999; Devictor et al. 2010; Bang and Faeth 2011; Corona et al. 2011) and improve management of human-wildlife conflict. Volunteer-science provides a large and robust pool of enthusiastic people interested in problem-solving and data collection. Furthermore, volunteer-science has facilitated analysis of ecological processes operating at broad spatial and temporal scales, far beyond the limit of traditional field studies (Wilson et al. 2013). Some of the largest wildlife-observation systems in the world rely primarily on volunteer effort to develop reliable, verified wildlife data (Schmeller et al. 2009; Ryder et al. 2010; Cooper et al. 2014). These volunteers are often professional biologists making wildlife observations in their free time and contributing these observations to various wildlife reporting systems (e.g. California Roadkill Observation System, CROS). One perception of volunteer science collected data is that they may suffer from observer bias and identification error (Cooper et al. 2014). However, this has not often been the case, and inaccuracies may be outweighed by the size of datasets available from volunteers (Schmeller et al. 2009; Ryder et al. 2010). As the volunteer science movement becomes an industry, it is anticipated that data collection will become more streamlined and standardized, with the volunteer scientist benefiting from the knowledge that they have helped advance in a scientific field they are passionate about. Informatics is a discipline that provides tools useful to collect, manage, and use diverse types of data to support research and management. Conservation-oriented analysis of ecological data collected by volunteers in standardized web-based informatics systems is a critical component of feedback to volunteers and can be an effective use of the data.

Volunteer and Agency Reporting of Road-Associated Wildlife

Globally, there are dozens of web-based systems for reporting WVC. For example, the Swedish National Wildlife Accident Council maintains a website for official reporting of accidents involving animals (http://www.viltolycka.se/hem/). The system is operated by the Swedish National Police, and it is the largest agency-owned WVC-reporting system in the world with over 200,000 records of WVC in the last five years. Online reporting and data display has been in place since 2010, but data from police records of accidents are available back to 1985. The largest, longest-running system that relies on volunteer-observers reporting any vertebrate species is the California Roadkill Observation System (CROS), maintained by the Road Ecology Center at the University of California-Davis (http://www.wildlifecrossing.net/california). In the US, the Idaho Department of Fish and Game operates the Idaho Fish and Wildlife Information System – IFWIS (https://fishandgame.idaho.gov/species/roadkill). The system allows entry of observation of any carcass resulting from WVC and as of 12/2014 had >22,000 records. Many observation systems have appeared over the last five years and they vary in their specific purpose, taxonomic breadth, and use of social networks for collecting data and outreach. A few use smartphone-based applications to facilitate data entry from the field (Olson et al. 2014) and some use social media and communication tools to receive observations (e.g., Project Splatter in the UK, http://projectsplatter.co.uk/). One purpose of this study was to find out whether it is possible to use the data from web-based informatics systems containing volunteer wildlife observations, to plan for WVC mitigation at the scale of US states.

Existing WVC reporting systems can consist of tens of thousands of data points and represent a potential source of “big data” for road ecology, community ecology, transportation mitigation, biodiversity mapping, and other scientific/engineering disciplines. Big data refers to datasets that are large and usually geographically extensive, and so require novel solutions for storage, analysis, processing and visualization (Hampton et al. 2013). At the US state scale and possibly at a global level, WVC reporting systems provide the largest known, continuous source of data on the occurrence and distribution of a wide taxonomic range of wildlife whilst also providing opportunities for tissue sampling of genetics, disease, and other testing. Carefully structured informatics (i.e. collection, storage, management, and sharing) systems for these observations facilitate ecological analyses and other biological uses of the data.

Spatial clustering of WVC

One common finding with spatial analysis of WVC is that collisions are clustered, which often leads to analysis of proximate causes of clustering for individual species (e.g., road or landscape features; Gunson et al. 2011). One approach is to use previous collisions to develop predictive landscape models to find "hotspots" (Nielsen et al. 2003; Langen et al. 2009; Gunson et al. 2011; Bil et al. 2013), or seasonality models to find "hot moments" (Beaudry et al. 2010). This is often done for ungulates because collisions with ungulates are both a conservation and safety concern (e.g., Danks and Porter 2010). There are various costs associated with a collision between a deer and a vehicle; on average, a collision with a deer costs $6,671 to society (Hujser et al. 2009). This approach means that WVC can be measured in terms of their cost to society, which can matter regardless of clustering of WVC. Less well-studied than WVC clustering is the idea that for broad taxonomic groups, "sheet flow" of animals may result in WVC everywhere and statistically-significant clustering may only be found because of limitations in the study area, or data collection. For highway planning, it is important to understand the clustering for individual species in each of their habitats and landscapes, and determine the reasons why higher WVC occurs on that stretch of road.

There are many tools to measure impacts to species from WVC, to determine causes and correlations with WVC, and for finding places where transportation agencies can focus remedial action to reduce impacts to wildlife and improve driver safety. Analysis to identify non-random clusters of single or multiple species WVC's (hotspots) has utilized GIS (Geographic Information Systems); a promising tool where statistics have been used to identify spatial clusters. Examples of analytical approaches and methods include: Nearest Neighbor Index (e.g. Matos et al. 2012); 'Satscan', borrowed from epidemiological studies, which looks for non-random clusters of events (i.e. disease outbreaks, Ball et al. 2008); the Getis-Ord- Gi statistic for spatial autocorrelation (Getis and Ord 1992); and the Kernel Density Estimator Plus method for estimating locations of high densities of events (Bil, personal communication).

We hypothesize that volunteer-collected observations of WVC could be used to prioritize roadway sections for mitigation action. We describe the use of data from state-scale, online observational networks for roadkill/wildlife occurrences in California (CA) and Maine (ME). We found that there were sufficient data to identify statistically-significant "hotspots" for many of the states' highways. We propose that novel online, volunteer-based systems could be used to augment the efforts of state DOTs and wildlife agencies and help inform location and type of mitigation actions.

Methods

We used a spatial-autocorrelation test (Getis Ord, Gi*) to determine the significance of WVC differences among neighboring roadway segments, where significance was set at $p < 0.05$. The two states were chosen for the availability of existing large-scale, online systems of volunteer-collected WVC data. At the time of writing, both systems were being actively used. The California Roadkill Observation System (CROS, http://www.wildlifecrossing.net/california) was launched in August 2009 to allow volunteer scientists to record carcass observations on California roads and highways. California has a population of more than 37 million people and >499,000 km of roadways networked across 411,000 km^2 of varied land cover types, including urban, agriculture, forests, grasslands, and desert. Of these roadways, 196,381 km are major roads, and 25,041 km are highways. Eighteen example highways were chosen in CA for geospatial analysis: interstates 5, 80, 280, and 580 and state routes (SR) 1, 3, 4, 13, 17, 20, 37, 49, 50, 70, 94, 99, 101, and 190. A similar system was developed in early 2010 for Maine, the Maine Audubon Wildlife Road Watch (http://www.wildlifecrossing.net/maine), to allow collection of both live and dead animal observations on and immediately adjacent to Maine's roads and highways. Maine has a population of 1,328,000 people and >60,600 km of roads, including 10,900 km of highways, across its 84,000 km^2 of forests, wetlands, agricultural areas and townships. Parts or all of 16 example highways were chosen in ME for geospatial analysis: interstate 29 and state routes 1, 2, 4, 7, 9, 16, 17, 100A, 111, 116, 126, 127, 128, 139, and 202.

WVC data collection

Volunteer-collected data were downloaded for each of CA and ME from their respective online systems. Date ranges for CA August, 2009 to October, 2014 and for ME were June, 2010 to November, 2014. WVC (n = 12,064) for specific highways were selected by hand in GIS based on their proximity to the highway. Any question about which of adjacent roadways a WVC was associated with was resolved by referring to the WVC record, which includes a narrative description of the site of observation.

The California Department of Transportation (Caltrans) maintains databases for carcass retrieval by District maintenance staff and for deer-vehicle-collisions (DVC) requiring a report and attendance by the California Highway Patrol. Partially-complete data-sets were retrieved from Caltrans using a request under the California Public Records Act. Data for portions of two Districts (3 & 4), were the most complete for carcass retrieval and accident reporting. Carcass retrieval data for 1984-1997 and 2001-2009 and DVC data for 2008–2010 were obtained for District 3, I-80 and SR50, and carcass/DVC data for 2005–2012 were obtained for District 4, I-280. DVC were summarized by tenth post-mile for each highway. Data from transportation-maintenance staff in Maine were not available at the time of the study.

Transportation management nexus: WVC hotspot analysis

Two types of "hotspot" analysis were conducted: a test for spatial autocorrelation, which identifies highway segments statistically-different from their neighbors, and calculation of WVC-density (# WVC/km-year), which allows comparison of WVC against some threshold of concern (Wang et al. 2010). These approaches are complementary in that there may be interest in high-densities regardless of whether or not clustering is statistically significant; conversely there may be interest in identifying geographically-discreet areas for mitigation action.

Each highway was dissolved into one long line segment and subsequently cut into regular-length segments of 0.40 km (0.25 mi) to 1.6 km (1 mi). These lengths were chosen because of previous research indicating that these are appropriate road segment lengths for studying wildlife crossings and WVC (Malo et al. 2004; Taylor and Goldingay 2004). WVC observations were forced into co-location with their respective highways using a "snap to line" tool (https://github.com/robintw/RTWToolsForArcGIS) implemented in ArcGIS 10.1. The "spatial join" tool in ArcGIS 10.1 was used to sum the number of observations per line segment and these sums per line segment length were used as the basis for density-based analyses and for subsequent spatial autocorrelation analysis.

Number of hotspots in California and Maine

We used a measure of spatial autocorrelation test called the Getis-Ord Gi* z-score statistic (Getis and Ord 1992) to determine whether or not WVC observations in California and Maine were spatially clustered in "hotspots" along highways. The Getis-Ord Gi* z-score is a measure of the statistical significance of clustering for each analysis unit, in this case highway segments. The Getis-Ord Gi* z-score was calculated using the default settings in ArcGIS 10.1.

Hotspot locations and spatial and temporal scales

Highway-specific observations were separated by year of observation, for full years of data: 2010, 2011, 2012, and 2013. Spatial autocorrelation of observations was determined for each year of observations. Different lengths of highway segment can affect where hotspots are identified. Shorter segment lengths (e.g., 0.4 km) may result in more hotspots than longer segments (e.g., 1.6 km) because there is greater likelihood at shorter distances that there will be a difference among segments in terms of # carcasses than at greater distances. The potential effect of varying highway segment lengths on hotspot identification was analyzed by carrying out autocorrelation analysis with 3 segment lengths: 0.4, 0.8 and 1.6 km.

Comparison of state agency and volunteer-collected data

Caltrans WVC data were used separately from volunteer-collected data from the California Roadkill Observation System (CROS) to analyze spatial autocorrelation and carcass density. Mule deer (*Odocoileus hemionus*) comprised >95% of Caltrans observations for many highways and were selected from all Caltrans data (carcass retrievals and collisions) to determine density of deer-vehicle-collisions (DVC) along select highways.

Cost of deer-vehicle collisions

We also used estimates of the total cost of deer-vehicle collisions to provide estimates of the cost per mile segment per year from deer-vehicle collisions (Hujser et al. 2009). Deer-vehicle collision data were from both Caltrans and CROS databases and were summarized to the tenth post-mile. There are various costs associated with a collision between a deer and a vehicle. On average, a collision with a deer costs US$6,671 (Hujser et al. 2009). We used this estimate of the total cost of DVC and segment-specific densities of DVC to provide estimates of the cost per mile segment per year from DVC. This provides another way to prioritize areas for mitigation, including both spatial location and economic benefits from mitigation action.

Results

Number of hotspots in California and Maine

The total number and length of statistically-significant clusters ($p < 0.05$), or "hotspots", were determined for highways and interstates in each of California and Maine (Table 1, Figure 1). Twenty-eight percent (6,940 km) of California's 25,041 km of state highways and interstates and 9% (947 km) of Maine's 10,900 km of state highways and interstates were analyzed for hotspots, though not all highways had WVC observations along their entire length. The length of individual hotspots varied considerably, from 0.8 km to 17.7 km. The total length of all hotspots increased significantly with length of highway analyzed ($p < 0.02$) at a rate of 10%, 0.10 km/km (Figure 2). If this rate held for all highways, the total length of hotspots would be 2,504 km in California and 1,090 km in Maine.

Hotspot locations and spatial and temporal scales

A few highways had sufficient data to conduct year-specific cluster analysis for 2010, 2011, 2012, and 2013. For one example highway, CA-49, certain hotspots persisted throughout the 4 years of data collection (Figure 3A) and the majority were present in one or several years (Figure 3A). For another highway, CA-13, locations of hotspots varied from year to year (Figure 3B). One example highway (CA-190) was segmented into varying-lengths for analysis, from 0.40 km to 1.6 km (Figure 3C). There was a tendency for shorter segments to result in a greater number of identified statistically-

Table 1. Statistically-significant clusters ("hotspots", $p < 0.05$) of dead animals (California, CA) and live and dead animals (Maine, ME) along state highways and interstates. The # of distinct hotspots and the total length of hotspots were determined for each highway.

Highway (length analyzed, km)	# observations/ observers	# observations/km	#/km Hotspots
CA-5 (1,283)	1,441/58	1.16	42/87
CA-50 (109)	415/18	3.81	7/42
CA-280 (39)	380/14	9.74	1/3.2
CA-80 (328)	679/50	2.07	7/24
CA-101 (1,302)	1,677/92	1.29	8/103
CA-99 (669)	350/37	0.52	3/40
CA-1 (1,053)	722/50	0.69	6/203
CA-49 (473)	540/37	1.14	4/82
CA-37 (35)	266/21	7.60	3/4.8
CA-4 (306)	217/21	0.71	3/19
CA-20 (341)	481/20	1.41	2/11
CA-3 (233)	309/8	1.33	1/85
CA-580 (122)	335/25	2.75	2/5.6
CA-13 (14)	580/7	41.4	2/2.0
CA-17 (43)	68/13	1.58	1/4.8
CA-70 (290)	617/60	2.13	12/28
CA-94 (56)	899/7	16.1	1/11
CA-190 (209)	637/12	3.05	3/31
(6,940)	10,612/ND		97/760
ME-295 (87)	394/30	4.53	3/8.0
ME-127 (24)	95/3	3.96	2/2.4
ME-116 (69)	45/1	0.65	1/0.8
ME-111 (22)	33/3	1.50	1/0.8
ME-128 (21)	60/4	2.86	2/2.4
ME-139/202/100A (40)	293/5	7.33	2/4.0
ME-17/126 (23)	51/4	2.22	0/0
ME-2/7/9 (37)	79/7	2.14	2/1.6
ME-4/16 (87)	107/6	1.23	2/5.6
ME-1 (537)	295/47	0.55	2/127
(947)	1,452/ND		17/153

significant clusters (0.4 km: n = 20 clusters) and longer segments to result in fewer and longer clusters (1.6 km: n = 4 clusters). For many of the highways, the statistically significant hotspots often overlapped at these different scales.

Comparison of state agency and volunteer-collected data

The vast majority of Caltrans observations were of mule deer (*Odocoileus hemionus*). For example, during one reporting period along I-80 (1967 to 1992), there were observations of 906 mule deer, 5 black bear (*Ursus americanus*), 1 beaver (*Castor canadensis*) and

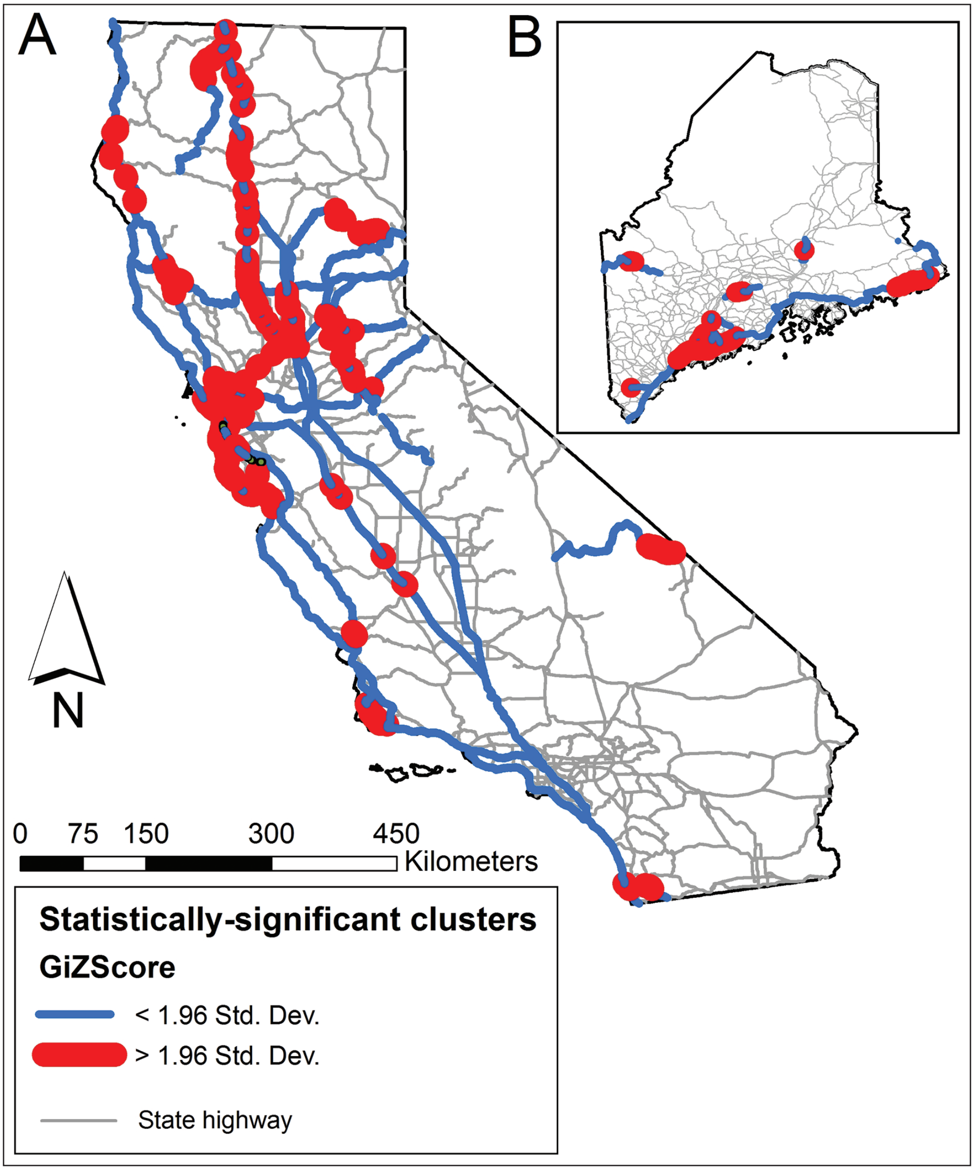

Figure 1. Locations of hotspots on California and Maine highways. The Gi* statistic, Z-score indicates the statistical significance of WVC clusters. A score of >1.96 indicates a statistically-significant cluster ($p <$ 0.05); scores lower than 1.96 are not significant ($p > 0.05$).

1 raccoon (*Procyon lotor*). This dominance of observations of deer is likely to be different for more urban areas. In comparison, observations from the CROS for I-80 (2009 to 2014) included 679 individuals from 63 species, with 69 individuals being mule deer. For the highways where state agency and volunteer-collected data were available, the carcass counts from each source for the most part did not overlap (Figure 4A, B). In other words,

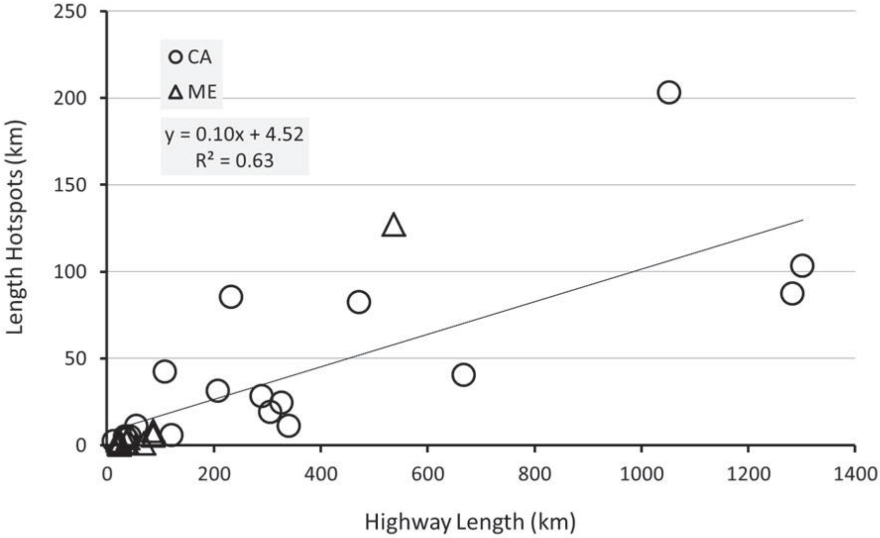

Figure 2. Relationship (CA and ME) between length of hotspots and highway length. The formulas and R^2 values are for the combined ME and CA data.

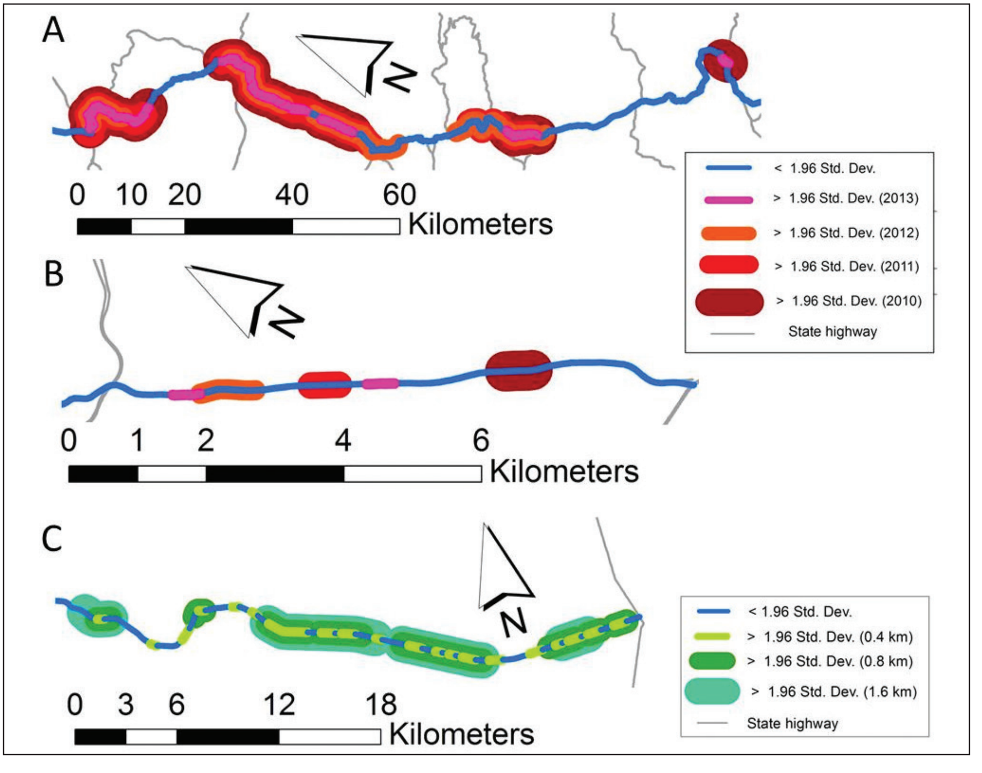

Figure 3. Geographic variation in hotspots among years and with varying highway-segment lengths. Annually-specific hotpots for **A** CA-49 and **B** CA-13 **C** Variation in position and number of hotspots along CA-190 with varying segment lengths: 0.4, 0.8, and 1.6 km segments.

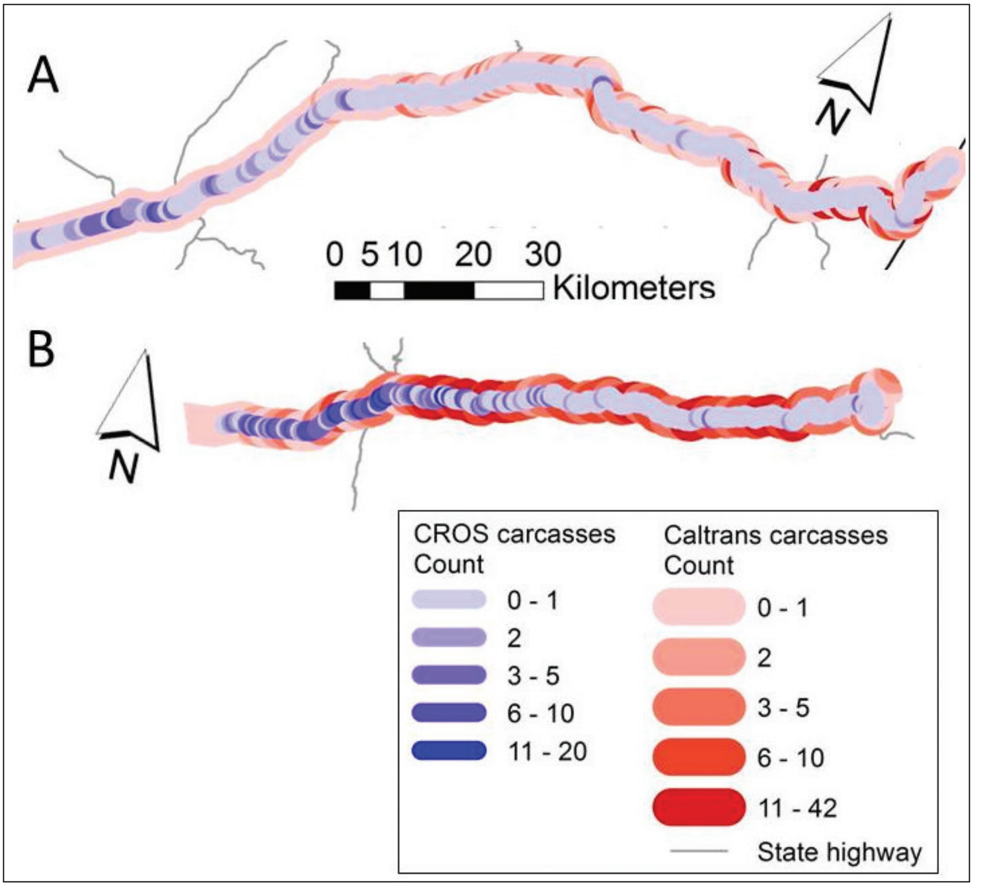

Figure 4. Comparison of state agency and volunteer-collected data-based hotspots. Carcasses reported in the CROS system (inner blue-range segments) overlaid with carcasses reported in the Caltrans system (outer red-range segments) along CA-80 (**A**) and along CA-50 (**B**).

where carcass counts from CROS were high, carcass counts from Caltrans were often low or nonexistent, and vice-versa. Similarly, the hotspots calculated using each source of data did not overlap with each other (Figure 4C, D). State agency data were dominated by mule deer carcasses, which were primarily collected at higher elevations and away from urban areas. Although data collection by volunteers also occurred in these areas, hotspots from their data were primarily identified near developed urban and agricultural areas.

Cost of Deer-Vehicle Collisions

Identifying locations of WVC clusters is one type of information useful for transportation mitigation planning. Identifying locations of high-cost from deer-vehicle collision (DVC) is another type. For one highway (CA-50), there was some overlap of hotspots

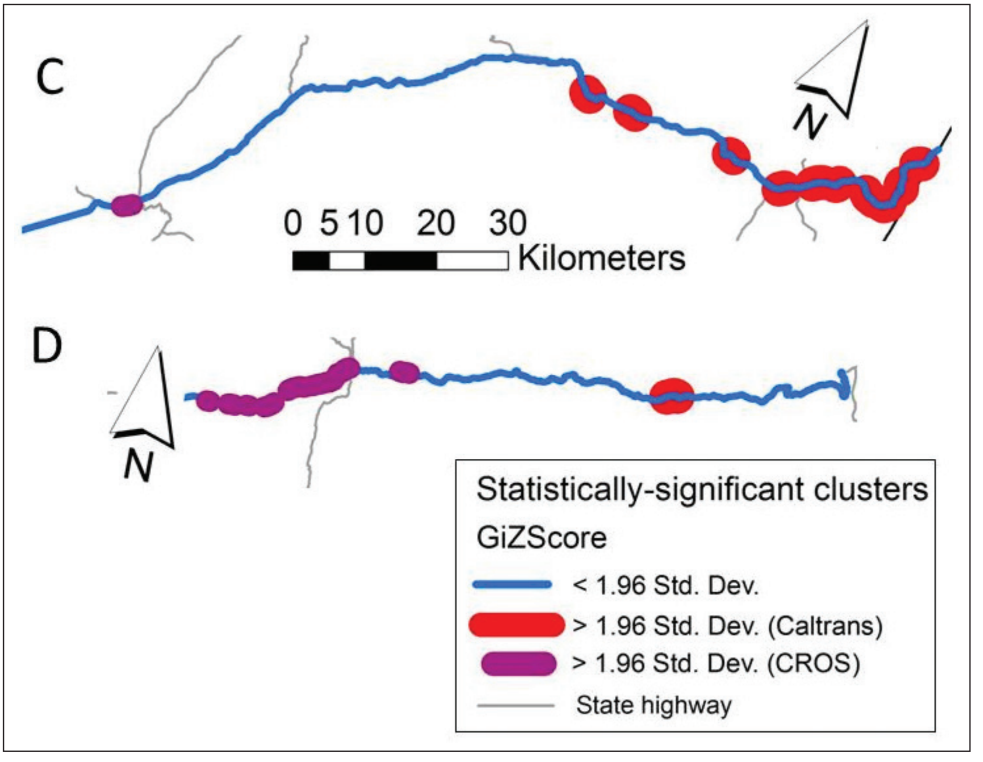

Figure 4. Continued. Locations of statistically-significant clusters from CROS data (purple segments) and Caltrans data (red segments) for CA-80 (**C**) and for CA-50 (**D**).

identified from volunteer observations of all species of WVC and 2 locations of high estimated cost of DVC from volunteer and DOT observations (Figure 5). For CA-50, the estimated annual cost of DVC ranged from <$500 to >$10,000 per mile (Figure 5). For another highway (I-280), according to Caltrans databases, there have been 362 collisions with deer between January, 2005 and July, 2012, or roughly 48/year. For I-280, there was very little overlap between the single hotspot identified from volunteer observations and the longer stretches of high estimated annual cost from DVC (data not shown). Also, the estimated annual cost of DVC was higher than for SR 50, reflecting a higher rate of DVC, and varied from <$1,000 to >$40,000 per mile.

Discussion

We demonstrate that volunteer observations of WVC from across a broad taxonomic range can be used in WVC hotspot identification on state highways. Within each of CA and ME, the systems described here represent the most extensive and taxonomically-broad wildlife monitoring effort, providing information about herpetofauna, birds, and

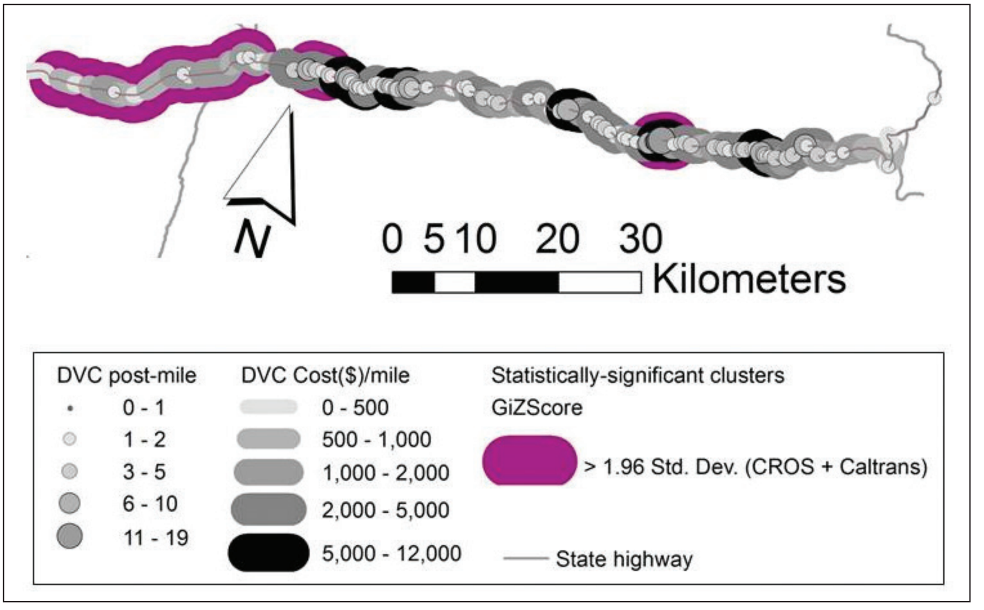

Figure 5. Locations of potential, cost-effective areas for mitigation. Statistically-significant clusters using volunteer observations (outer purple segments), annual rate of DVC per post-mile (points), and associated costs ($/mile, gray segments) of DVC for CA-50.

mammals. The opportunistic wildlife observations in our systems may provide the raw data for statistical analyses of proximate contributors to wildlife-vehicle collisions and planning for minimizing WVC impacts on wildlife and drivers. Targeted surveys could be used to understand the impact of WVC on local wildlife populations, a critical need in understanding and mitigating transportation impacts (Fahrig and Rytwinski 2009).

We demonstrate here that a network of volunteer observers at the US state-scale provide information potentially-useful to DOTs in planning mitigation. In ME, records of all wildlife observations from 2012 were shared with Maine Audubon's project partner the Maine Department of Transportation (MDOT) for use in their project scoping process (Maine Audubon, personal communication). Maine Audubon plans to continue annually to provide them with all observations as well as results from hotspot and density analysis (Maine Audubon, personal communication). The plan is to identify where areas of conservation concern overlap with MDOT projects in their 3-year plans. Where there is overlap through assessment of the habitats, species types, and road characteristics, projects can be designed to mitigate impacts to wildlife and public safety and enhance wildlife movement. In addition, locations of hotspots and high density of live and dead wildlife observations will be shared with local volunteer science volunteers for them to share and work with their towns planning and road departments for local road project mitigation. We hope that a similar DOT use of our hotspots analysis will also occur in California.

Wildlife-Vehicle Collisions

Animals die as result of collisions with vehicles because of traffic speed, traffic volumes, seasonal changes in movement, separation of important habitat areas, occluded line-of-sight, and other factors (Barthelmess 2014; Hobday and Minstrell 2008; Litvaitis and Tash 2008). Most of the observations of dead animals made using the online, state systems described here were opportunistic and thus do not reflect actual rates of WVC on a particular roadway. For certain highways analyzed in the present study (CA-13, CA-190, CA-94, I-280) known observers have consistently and frequently made observations of WVC, thus in these cases the reported rates are a closer approximation of actual rates, especially for larger animals that are both easily observed and more difficult for scavengers to displace. WVC may occur and not be observed, be removed by highway maintenance crews, or be scavenged by other animals. Scavenging rates can be very high for roadkilled animals, affecting confidence in estimates of total impact of WVC on populations (Antworth et al. 2005; Barthelmess and Brooks 2010).

The observations in the current study do reflect the presence of particular species at particular times of year and thus are a presence-only type of record. These data are useful in understanding wildlife distribution and movement, and for roadkilled animals, proximate causes of the collision (Barthelmess 2014) or, as demonstrated here for frequently-driven roads, spatial-aggregation of collisions. Large-extent databases of WVC observations provide a tool for developing and testing predictive models for contributing factors to WVC. Because of unevenness in sampling and the unknown level of effort going into opportunistic reporting in the systems described here, we are not in a position to rank risks to wildlife among highways. However on single routes with high and/or regular rates of observation, local hotspots (blind curves, riparian crossings) may be located and calibration made of observations per unit effort, relative visibility and reporting rates found for different species, and other bias-correction rates calculated.

Mitigation planning

We demonstrate that volunteer-observations of WVC can contribute to understanding locations of WVC clusters that could be suitable for mitigation action at US state scales. We found that the length of highway segments analyzed had an effect on the position and occurrence of clusters. This is similar to the finding for bird species richness, where geographic clustering was found to depend on analytical scale (Ma et al. 2012). Because of this, the best approach for mitigation planning would be to carry out cluster analyses for multiple segment lengths, depending on the taxonomic group or process of interest. Previous research indicates that road segment lengths of 0.4 to 1.6 km area appropriate for studying wildlife crossings and WVC (Malo et al. 2004; Taylor and Goldingay 2004). Representation of cluster locations across multiple segment length classes may indicate places of particular importance from a collision point of view. For future studies, it would be worth formalizing segment-analysis lengths that reflect a combination of consideration of ecological processes (e.g., species-dependent movement distances) and transportation-planning (e.g., segment scales for planning).

Identifying locations of clusters of WVC is a common step preceding mitigation and conservation actions to protect wildlife from vehicle-caused mortality (e.g., Hobday and Minstrell 2008). In the present study, cluster locations were found to vary in position along study highways across years of observations. For one highway (CA-13), there was virtually no overlap among WVC clusters from year to year (2010 to 2013). For another highway (CA-49), there were locations where clusters were identified every year, from 2010 to 2013. There were also locations where clusters were identified during 1, 2, or 3 of the 4 years. It is likely that factors contributing to WVC, such as traffic volume and speed, land cover, and road characteristics, did not change significantly during the study period. This suggests that temporal-dependence of cluster-locations is related to changes in the behavior of individuals and species along these highways. In addition, locations of statistically-significant clusters are not the only locations for concern about WVC. Highways with high rates of WVC across many adjacent segments may have few clusters, but many areas of concern because of impacts to drivers and animals (e.g., Figure 4, CA-50). This type of finding is very important for conservation planning, because it suggests that there may not be predictable landscape "corridors" or "linkages", with corresponding stretches of highway suitable for mitigation action to protect wildlife movement. This finding contrasts with previous findings for certain taxonomic groups. For example, Langen et al. (2008) found that locations of clusters of herpetofauna road mortality were stable over time (i.e. comparison of 2002 and 2006/07). These clusters co-occurred with ponds and wetlands, which could explain the lack of change over time. We did not have sufficient data to divide the WVC observations into individual species and years.

The hotspots identified from volunteer-observations may not align with clusters identified using Department of Transportation (DOT)-collected WVC observations, because the latter are typically of ungulate and other large species that pose a risk to drivers. The combination of high-species-diversity observations by volunteers and DOT/wildlife agency observations of large animals could provide the ideal combination of WVC data to directly inform mitigation planning that provides both conservation and driver-safety benefits. In addition, because of the taxonomic breadth of volunteer-collected WVC observations, individual species could be considered for safety (e.g., mule deer) or conservation (e.g., meso-carnivores) reasons.

The annual cost of deer collisions, varied between the two CA state-highways analyzed and ranged from <US$ 500 to >US$30,000 per mile. To put these numbers in perspective, it can cost ~US$25,000/mile to augment a 5-6 foot chain link fence to make it into an 8-foot, deer-resistant fence (e.g., deer-fence in ID, https://fishandgame.idaho.gov/content/post/i-15-mule-deer-fence-near-pocatello-complete) and up to US$100,000/mile to construct a new 8-foot, deer-resistant fence. Fences are typically associated with purpose-built crossing, or other, structures that allow wildlife passage acros a right-of-way. There were segments of high costs from deer collisions

(>US$5,000/mile-year) throughout both SR 50 and I-280. Fence/crossing mitigation of certain stretches of state highway could pay for themselves in terms of avoided costs from deer collisions in a matter of 1–20 years, depending on rate of collision and existing fence infrastructure.

Many segments of the state highways studied are likely to have collisions between vehicles and any animal, including deer. These areas may or may not be predictable, but what is certainly predictable is that providing directional fencing to encourage deer and other wildlife to usable crossing structures will reduce WVC. Directional fencing and accompanying escape structures (e.g., jump-outs to allow animal escape from the road-side of a fence) and highway under or over-crossings have proven to be effective for reducing collisions between wildlife and vehicles (Hedlund et al. 2004; Seamans and Helon 2008). This utility is predictably compromised if the structures and materials are not monitored and maintained causing more animals to enter the roadways. At the scale of whole states and state highways, these structures will seem expensive, though not compared with the costs in lives, injury and property damage from collisions, or swerving to avoid collisions, with animals. Thus, strategically placing mitigation structures and showing their potential and actual cost-effectiveness will be very important for a more wide-spread adoption.

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RESEARCH ARTICLE

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Testing alternative designs for a roadside animal detection system using a driving simulator

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Abstract

Objectives: A Roadside Animal Detection System (RADS) was installed in January 2012 along Highway 41 through Big Cypress National Preserve in Florida, USA in an attempt to reduce wildlife-vehicle collisions. The system uses flashing warning signs to alert drivers when a large animal is near the road. However, we suspected that the RADS warning signs could be ignored by drivers because they resemble other conventional signs. We hypothesized that word-based warning signs (current design) are less effective than picture-based signs at catching drivers' attention. **Methods:** We used a driving simulator to test (1) the effects of the RADS on collision rate, driver speed, and latency to brake; and (2) whether the RADS would be more effective if warning signs were picture-based. Participants were randomly assigned to one of three treatments: no warning (control), word-based RADS signs (current design), and picture-based RADS signs (proposed design). During the simulations, a deer entered the road in front of the driver, and we recorded whether drivers "crashed" or not. **Results:** Both the picture-based and word-based RADS signs resulted in significantly lower crash probabilities. The picture-based RADS signs performed better than the word-based signs in reducing speed and latency to brake, although the effect varied between twilight and night. However, the word-based RADS signs still did produce significant reductions in speed and braking latency. **Conclusions:** We conclude that the word-based RADS in Big Cypress should help prevent dangerous wildlife-vehicle collisions, but that redesigning the warning signs to be picture-based could yield even greater benefits.

Keywords

Animal detection system, animal-vehicle collisions, driving simulator, traffic safety, RADS

Introduction

Collisions between large mammals and vehicles are costly for wildlife and humans alike. In Sweden, road-kills are responsible for an average loss of between 1 and 12 percent of the population size of medium- and large-sized mammal species (Seiler et al. 2004). Road-kill disproportionately affects small reptiles, amphibians, and mammals, but collisions with large mammals are most frequently reported (Huijser and McGowen 2003). This is because collisions with large mammals come at a greater cost to humans; in the United States, they cause 211 deaths, 29,000 injuries, and more than $1 billion USD in property damage every year (Huijser et al. 2007). Likewise, large animal-vehicle collisions (hereafter LAVCs) take a heavy toll on wildlife populations, through increased mortality and reduced landscape connectivity. In fact, roads are thought to be one of the greatest threats to wildlife worldwide (Noss and Cooperider 1994; Trombulak and Frissel 2000; Forman et al. 2003; Smith 2003; Laurance et al. 2014).

The most common measures used to reduce the incidence of LAVCs are static warning signage, wildlife fencing and ecopassages (e.g., overpasses, underpasses, and tunnels/culverts). The first, static warning signage, has been shown to be largely ineffective; drivers easily habituate to it and fail to make adequate reductions in speed (Huijser et al. 2007). The other two measures are largely successful, but they come with limitations. On its own, wildlife fencing merely creates another barrier to animal movement throughout the landscape (Smith 2003), so it is rarely a standalone mitigation method for large, wide-ranging animals, though it can be a useful interim measure (Jaeger and Fahrig 2004). Using fences in combination with ecopassages is a common strategy, but the effectiveness of ecopassages is almost never evaluated in a BACI design (van der Ree et al. 2007; Lesbarrères and Fahrig 2012) and they can be very expensive (several million USD; Huijser et al. 2007). In addition, their installation is disruptive to traffic, meaning that they are rarely installed unless a road is being widened or a new road built (Smith 2003; Huijser et al. 2007). Because of the cost and difficulty of installation, ecopassages cannot be installed at every location that could benefit from one.

Roadside Animal Detection Systems (RADS) are a relatively new alternative to the previously listed measures. RADS use sensors (e.g. motion-sensing, thermal, infrared) to detect when large animals are near the road; when an animal is detected, the sensors send a signal to lights on warning signs, which begin to flash. Unlike fencing and ecopassages, RADS are not intended to keep wildlife off of the road, but rather to alert drivers when there is an increased risk of a collision. Because the lights only activate when a large animal is detected near the road, it should decrease the likelihood of drivers habituating to the warning signs. In addition, RADS are easier to install than ecopassages and are less expensive, ranging from $11,500–$60,000 USD plus installation and maintenance costs (Huijser and McGowen 2003), so they could potentially be deployed on a larger scale. RADS were first installed and tested in 1993 in Switzerland (Kistler 1998, Tschuden 1998; cited in Huijser and McGowen 2003). Since then, many more systems have been implemented in North America and in Europe:

in 2006, there were 34 separate locations with RADS installed (14 inactive), with 27 more planned (Huijser et al. 2006).

Despite the promising nature of the technology, the success of a particular RADS depends largely on its ability to influence human behavior. However, studies to evaluate driver response to RADS in the field have encountered significant difficulties (reviewed in Huijser and McGowen 2003). For example, false triggers—instances where the system activates when no large animal is present, often caused by vegetation or deep snow—have been a common problem. Over time, this could cause driver habituation and skew the results of a field evaluation. Broken sensors and loss of power have also been an issue in certain locations, especially where weather conditions are severe, e.g., excess rain or snow. As a result of these technical challenges, it is hard to draw conclusions from previously collected data; however, one study of a RADS at a deer crossing that did not experience malfunctions until after the study period found that the system reduced driver speed by 6.5 km/h at night and 3 km/h during the day (Gordon et al. 2004).

We circumvented these difficulties by using a driving simulator to evaluate how a RADS affects driver behavior. Our study is the first to use a driving simulator to assess a RADS; however, simulator studies have been used to assess driver reaction to variations in road signs, e.g. Hammond and Wade (2004). The use of a driving simulator is ideal for evaluating RADS in many ways: first, it provides a controlled setting in which to observe driver behavior, with no confounding effects of weather, time of day, or equipment malfunction. Second, it allows us to safely assess the risk of an animal-vehicle collision; in a simulated setting, this can be done with no threat to humans or wildlife, as well as generating a much larger sample of potential collisions than could be observed in the field.

Another key feature of the simulator is that the controlled setting allows us to compare alternative designs for the RADS; in this case, designs for warning signage. Bond and Jones (2013) show that drivers rate warning signs with silhouettes of animals (picture-based) higher than text-only signs (word-based) in their ability to reduce speed and increase alertness. Both picture- and word-based signs have been used in RADS systems (reviewed in Huijser and McGowen 2003); however, the two alternatives have never been compared in a controlled setting. It is possible that one design could produce the desired reduction in speed and/or increase in alertness more often or with a greater magnitude. For identifying best practices in animal-vehicle collision mitigation, we sought to determine which design yields greater potential benefits.

In this study, we asked (1) does the presence of a RADS have an effect on driver speed, latency to brake when an animal enters the road unexpectedly (treated here as a proxy for alertness), and/or the probability of animal-vehicle collisions? and (2) do word-based and picture-based RADS signs perform differently? Because the use of a driving simulator removes the technical challenges often experienced in the field, we can evaluate whether RADS have the potential to significantly affect driver behavior and reduce crashes with wildlife. If so, it will be worthwhile to invest in research and development to overcome these challenges.

Methods

Focal RADS

In January 2012, an experimental RADS (manufactured by Simrex Corporation) was installed in Big Cypress National Preserve (BCNP) on Highway 41 near Turner River in Collier County, Florida, USA. Daisy-chained infrared sensors, spaced approximately 153 m apart and placed just beyond the road shoulder on both sides of Highway 41, create a detection beam parallel to the road surface spanning 2.1 km. The system is designed specifically to detect large wildlife, so the infrared beam is 46 cm above the ground and will not detect shorter animals. The standard yellow diamond warning signage is word-based, not picture-based, and reads, "WARNING WILDLIFE ON ROADWAY REDUCE SPEED." These warning signs are placed every 610 m. In addition, an informational sign informs drivers of the RADS system 0.8 km before entering the animal detection area (Fig. 1).

The 2.1-kilometer-long road segment was identified by federal and state wildlife agencies as a critical hotspot for road-kills of the federally endangered Florida panther (*Puma concolor coryi*), whose population is currently estimated at 100–180 individuals (FFWCC 2014). Five panther road-kills occurred at this location between 2004 and 2009, of which four were breeding-age females. Although collisions with Florida panthers were the catalyst for this experimental installation, a successful RADS would also benefit other large wildlife, such as Florida black bear (*Ursus americanus floridanus*), and white-tailed deer (*Odocoileus virginianus*). Since its installation, the RADS has experienced a large number of false triggers during daylight hours, which could potentially reduce its effectiveness as a warning system.

Driving simulator

The driving simulator was provided and programmed by the Research in Advanced Performance Technology and Educational Readiness Lab at the University of Central Florida's Institute for Simulation and Training. They programmed the simulator to create a digital version of the RADS installation site on Highway 41. Using this digital version of the roadway and surroundings as a base, three alternatives were created: one had no RADS warning signs (control), another included word-based RADS signs (which reflects the Highway 41 RADS), and the third included redesigned, picture-based RADS warning signs (Fig. 1).

Participants

The use of human subjects in this research was approved by the UCF Institutional Review Board, IRB number SBE-13-09322. Ninety people participated in the simulator experiment between 15 July and 18 August 2013. The participants were either students

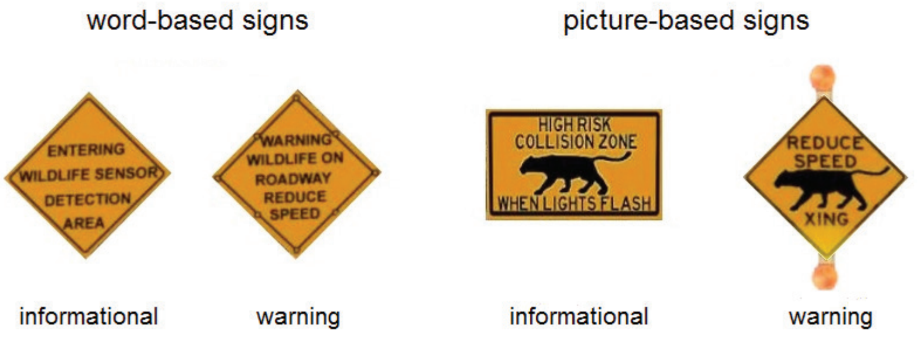

Figure 1. Images of the RADS signs used in the driving simulator. Left: simulated images of the Highway 41 RADS signs, including a word-based informational sign posted 0.8 km before entering the animal detection area and four word-based warning signs with 8 LED lights that flash when animals break the infrared beam parallel to the road. Right: simulated images of the modified RADS signs, which feature a picture-based informational sign and four picture-based flashing warning signs.

or employees of the University of Central Florida or members of the surrounding community. Undergraduate students were offered course credit as incentive to participate, while all other participants received a small monetary compensation. All were at least 18 years old, and all had been licensed to drive for at least one year.

Participants were recruited systematically so that there were 30 participants in each age group (age group 1 = 18–24 years, age group 2 = 25–44 years, age group 3 = 45+ years). We recruited into these age groups to obtain a more balanced participant pool, and because age is known to affect driver behavior: young or inexperienced drivers make up a disproportionate amount of accidents on the road because they have an underdeveloped ability to recognize hazards, yet tend to overestimate their own driving skills (reviewed in Deery 2000).

Participants in each age group were assigned to a treatment (control, word-based RADS, or picture-based RADS) using a systematic random design: i.e., the first member of each age group tested was assigned to the control treatment; the second, word-based; third, picture-based; fourth, control, *etc*. Thus, exactly one-third of each age group was assigned to each treatment.

Participants were told that they were participating in a study aiming to evaluate driver response to various hazards on the road. The full nature of the study, i.e., the intent to evaluate the RADS, was not disclosed to the participants until debriefing after the testing session. This was done so that participants did not anticipate seeing animals on the road, which could affect their responses.

Simulator experiment

Each participant completed six runs in the driving simulator, three "twilight" scenarios and three "night" scenarios. We chose these times of day because these are the times that collisions with large animals are more likely to occur (Danks and Porter 2010; Neumann et al. 2012), and also because in Big Cypress National Preserve, there is a nighttime speed limit of 45 mph (72 km/h), while during the day (and twilight) the speed limit is 60 mph (97 km/h). This was reflected in the speed limit signs programmed in the simulation. Before participants began the six runs, there was an acclimation period during which the participant was able to familiarize themselves with and become accustomed to the driving simulator. A five-minute break was offered between each run, during which the participant could walk around, get water, or eat.

In each set of three runs, there was one target run featuring an animal hazard in which a deer entered the road directly in front of the car at a certain point in the run. The other two were non-target runs, meaning that they did not include an animal hazard, but instead a different type of hazard (either a car crashed on the roadside or a driver entering the road in front of the participant suddenly) to prevent the participant from realizing the true purpose of the study. Each run featured only one hazardous situation. Participants completed the six runs in a random order.

Each run featured a 0.8 km baseline period at the beginning, a 0.8-km zone after the RADS informational sign, and a 2.1 km RADS zone. All hazards, animal or not, occurred within the RADS zone. Driver speed and brake pressure were automatically recorded every 0.014 seconds. We also recorded whether or not a participant crashed into the animal hazard during target runs.

Analyses

We only analyzed the target runs, in which participants were presented with an animal hazard. Each participant completed one target run during the twilight scenario and one during the night scenario. Because each participant was tested both at night and twilight, which had different speed limits, the effects of these factors were tested using paired methods. All statistical tests were performed in JMP® (version 10, SAS Institute Inc., Cary, NC, USA) and figures generated using JMP® or R Statistical Computing software (version 3.0.2, R Foundation for Statistical Computing, Vienna, Austria, 2013).

Pairwise comparisons between twilight and night

To assess whether there was a significant difference in crashes between runs that occurred at twilight vs. those at night, we performed McNemar's chi-squared test for paired samples. To test whether driver speed was greater at twilight than at night, we used Wilcoxon's signed rank test (1-tailed). The speed considered was the average speed between the point where the participant entered the RADS sensor array and the point just before the deer appeared in the road. We used a nonparametric test because the differences between the pairs were not normally distributed (Shapiro-Wilk test, W = 0.953, p = 0.00260). To test if there was a difference in latency to brake (measured as the distance between the location where the participant started braking and the location of the deer on the road) between night and twilight, we again used Wilcoxon's signed rank test (1-tailed) (Shapiro-Wilk test, W = 0.945, p = 0.00131). We excluded data from six participants who did not brake in response to the animal hazard in one or both of their target runs: four participants from the control treatment, one participant each from the word- and picture-based treatments; three participants from age group 1 and three participants from age group 2.

Effect of RADS

To assess whether treatment or age had an effect on the probability of crashing, we fit a multiple logistic regression model using "crash" (yes/no) as the dependent binomial outcome and treatment (control, word-based RADS, and picture-based RADS) and age group as fixed factors. We intended to do this analysis for both twilight and night, but so few crashes occurred at night (n = 2) that the logistic model becomes unstable and uninformative. Therefore, this analysis was done only for twilight.

We used One-Way Analysis of Variance (ANOVA) to assess whether treatment or age had an effect on speed or latency to brake (measured as the distance between the location where the participant started braking and the location of the deer on the road). Separate ANOVAs were done for twilight and night datasets.

Results

Descriptive statistics

We tested 43 females and 47 males. Mean age (yrs.) and standard error in age group 1 was 20.4 ± 1.98 (range 18–24); age group 2, 30.3 ± 4.45 (range 25–41); age group 3, 52.2 ± 5.62 (range 45–65). Of the 90 participants, 30 crashed (Table 1).

Table 1. Number of crashes per factor.

Factor	Number of crashes			
	Control	Word-based	Picture-based	Total
Treatment	18	8	4	30
	Age group 1	Age group 2	Age group 3	Total
Age group	18	6	6	30
	Twilight	Night		Total
Time of day	28	2		30
	Males	Females		Total
Gender	13	17		30

Pairwise comparison of crashes occurring in twilight vs. night runs

There were 30 total crashes observed in our experiment. During twilight runs, participants crashed 31% of the time (n = 28 crashes), while in night runs, participants crashed 2% of the time (n = 2 crashes). This difference is significant (McNemar's χ^2 = 22.3214, DF = 1, $p < 0.00001$; Table 1). Thus, we analyzed the effect of experimental factors separately for twilight and night. However, since only 2 crashes occurred at night, we were only able to further analyze crashes that occurred in twilight simulations.

Effect of RADS on crash rate

The overall model for crash probability was highly significant (χ^2 = 24.9, DF = 5, $p < 0.0001$). The independent variables treatment and age group had a significant influence on crash likelihood (respectively, χ^2 = 17.5, DF = 2, p = 0.0002; χ^2 = 7.08, DF = 2, p = 0.0290; effect likelihood ratio tests).

Between levels of experimental factors, participants in the youngest age group (age group 1) were significantly more likely to crash than those in the oldest age group (age group 3; Tables 1 and 2). There was no significant difference in crash rate between age groups 1 and 2 or 2 and 3. Participants in the control treatment were significantly more likely to crash than those in the word-based treatment or the picture-based treatment (Tables 1 and 2). There was no significant difference in crash rate between the word-based treatment and the picture-based treatment.

Pairwise comparison of speed in twilight vs. night runs

Mean speed of participants during twilight runs was 93.7 km/h ± 1.21 SE and ranged from 70–160 km/h, compared to a nighttime average of 78.2 km/h ± 1.24 SE with a range of 23.0–160 km/h. Extreme outliers (>3 standard deviations from the mean) were seen in both twilight and night and were from the same participant; these were removed from further analyses. The average difference in speed between twilight and night runs for a participant was 15.4 km/h ± 1.09 SE faster at twilight than night, with a range of -20.4–34.1 km/h. This difference is significant (Wilcoxon signed rank test with continuity correction, W = 120, $p < 1.0e{-13}$).

Effect of RADS on speed

The overall twilight model for speed was significant (F = 2.55, DF = 8 and 80, p = 0.0159). Both treatment and age group had a significant influence on speed, and there was no interaction between treatment and age group (Table 3). Between the different treatments, participants in the control treatment (mean speed 97.0 km/h) drove faster than those in

Table 2. Crash probabilities between treatments and age groups. Odds ratios showing likelihood of a member of the first group crashing compared to the likelihood of a member of the second group crashing. A dagger (†) indicates significant odds ratios at α = 0.10 while an asterisk (*) indicates effects significant at α = 0.05.

Comparison	Odds Ratio	$p > \chi^2$	Lower 95%	Upper 95%
age group 1 vs. age group 2	3.54	0.0514†	0.993	14.1
age group 1 vs. age group 3	5.15	0.0114*	1.43	21.5
age group 2 vs. age group 3	1.46	0.591	0.36	6.06
control vs. word-based	6.29	0.0026*	1.85	25.4
control vs. picture-based	14.0	<0.0001*	3.59	68.9
word-based vs. picture-based	2.22	0.267	0.546	10.1

Table 3. ANOVA table for average speed of participants within the RADS zone at twilight. An asterisk (*) indicates effects significant at α = 0.05.

Factor	DF	SS	F ratio	p > F
age group	2	211.4	3.19	0.0464*
treatment	2	327.0	4.94	0.00951*
age group*treatment	4	136.8	1.03	0.396

the picture-based treatment (89.5 km/h), but not the word-based treatment (92.2 km/h) (Tukey-Kramer HSD, p = 0.0098 and 0.137, respectively; Fig. 2). Participants in age group 3 drove significantly slower than those in age group 2 at α = 0.1 (Tukey-Kramer HSD, p = 0.0560) but there were no other significant differences between the age groups (Fig. 2).

Nighttime speed data were transformed using the Box-Cox transformation to meet the assumption of normal residuals. The overall nighttime model was not significant (F = 1.75, DF = 8 and 80, p = 0.0991), and no factor significantly affected speed.

Pairwise comparison of braking distance in twilight vs. night runs

Mean braking distance during twilight runs was 45.7 m before the deer's location on the road (±1.99 SE; range 14.4–79.8), compared to 51.7 m during night runs (±1.81 SE; range 22.2–79.8). The average paired difference between night braking start distance and twilight braking start distance was that participants braked 5.93 m earlier at night than twilight (±2.80 SE), though there was a very wide range (-57.4 m–65.1 m). This difference is significant (Wilcoxon signed rank test with continuity correction, W = 2350, p = 0.00274).

Effect of RADS on braking distance

Both the twilight and nighttime overall models were significant (F= 3.77, DF = 8 and 75, p = 0.0009 and F = 3.81, DF = 8 and 78, p = 0.0008, respectively; nighttime

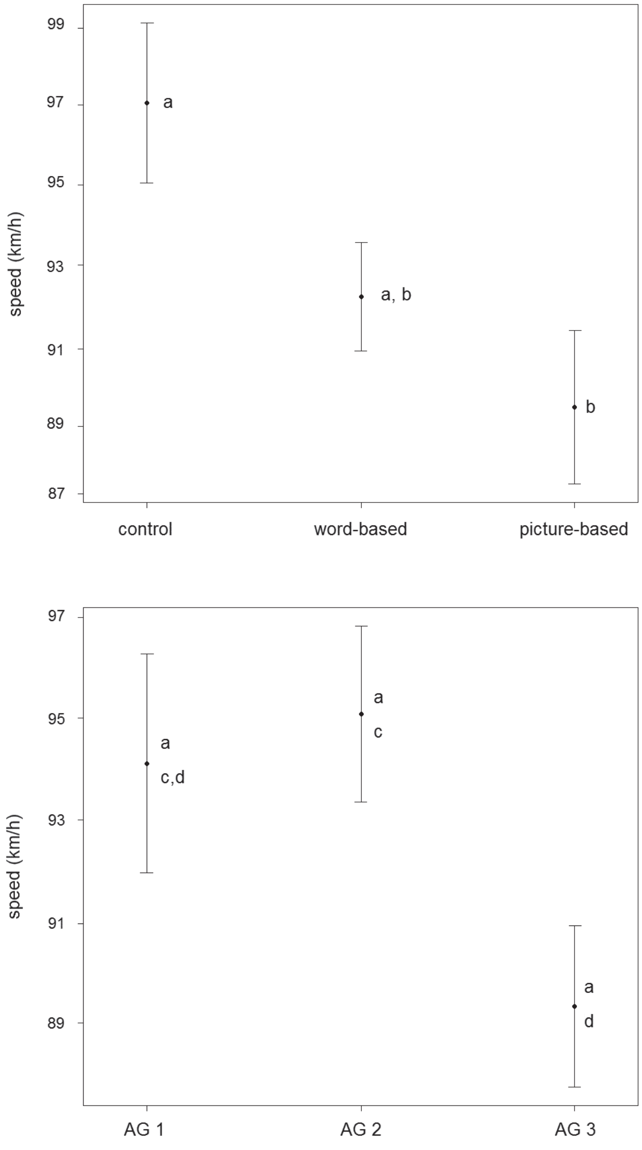

Figure 2. Participant speed at twilight by treatment and age group. Means and standard errors of speeds (mph) of participants at twilight in: **Top**- the three treatments; **Bottom**- the three age groups. Significantly different treatments have different letters next to them (a's and b's indicate significant differences at α= 0.05; c's and d's indicate significant differences at α= 0.10). n=30 for word- and picture-based treatments; n= 29 in control because we removed an outlier (>3 standard deviations from the grand mean).

braking distance data were transformed using the Box-Cox transformation to meet the assumption of normal residuals). At twilight, braking distance was influenced by treatment, though the effect of age group was also significant at α = 0.1 (Table 4). There was no significant interaction between treatment and age. Participants in both the pic-

Table 4. ANOVA tables for braking distance. The distance considered is the distance at which participants began to brake when the deer ran out in front of them during the simulation. Separate ANOVAs were calculated for twilight and night data. A dagger (†) indicates effects significant at α= 0.10 while an asterisk (*) indicates effects significant at α= 0.05.

time of day	factor	DF	SS	F ratio	p > F
twilight	age group	2	331.9	2.91	0.0605†
	treatment	2	1121.8	9.85	0.0002*
	age group*treatment	4	252.9	1.11	0.358
night	age group	2	110.4	2.16	0.122
	treatment	2	426.5	8.36	0.0005*
	age group*treatment	4	236.9	2.32	0.0643†

ture-based and word-based treatments began to brake earlier than participants in the control treatment (on average, 8.09 m and 7.59 m earlier, respectively; Tukey-Kramer HSD, p = 0.0007 and p = 0.0017, respectively; Fig. 3). There was no significant difference in braking distance between participants in the picture-based and word-based groups (Tukey-Kramer HSD, p = 0.968; Fig. 3).

At night, braking distance was influenced by treatment, but not age, although the interaction between treatment and age was significant at α = 0.10 (Table 4). Participants in the picture-based treatment started to brake on average 5.53 m before participants in the control treatment (significant difference; Tukey-Kramer HSD, p = 0.0006; Fig. 3) and 3.04 m before participants in the word-based treatment (significant at α =0.1; Tukey-Kramer HSD, p = 0.0725; Fig. 3). Participants in the word-based treatment started to brake 2.49 m before participants in the control treatment, but this difference was not significant (Tukey-Kramer HSD, p = 0.189).

Because the interaction between age and treatment at night was close to being significant at α = 0.05, and because we tested a relatively small sample of participants, we investigated the interaction for any trends indicating that the different RADS designs affected participants differently within an age group. Within age group 1, the youngest age group, those in the picture-based group braked on average 7.76 m before those in the control group (Tukey-Kramer HSD, p = 0.0749). Within age group 3, the oldest age group, those in the picture based group braked on average 7.81 m before those in the word-based group (Tukey-Kramer HSD, p = 0.0714).

Discussion

In our simulator study, we found that the RADS produced significant positive outcomes: participants in RADS treatments reduced their speed, braked earlier in response to an animal in the road, and were involved in animal-vehicle collisions less often. In addition, the picture-based RADS signs were more effective than the word-based RADS signs at reducing driver speed at twilight, and the same was true of differ-

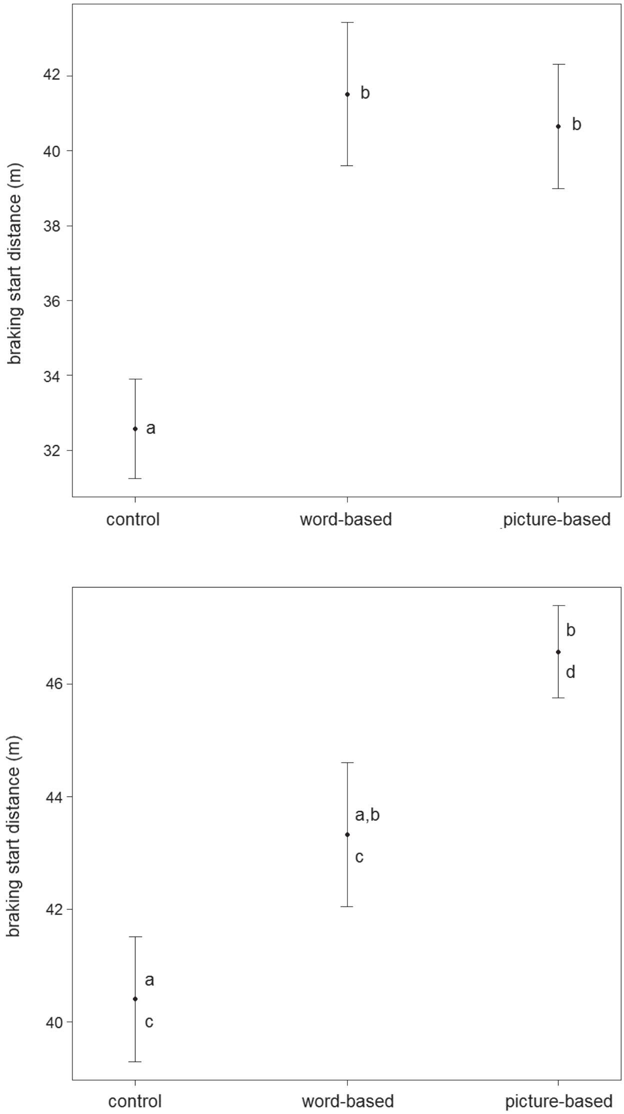

Figure 3. Braking distances by treatment. Means and standard errors of braking distances (m). Significantly different treatments have different letters next to them (a's and b's indicate significant differences at $\alpha = 0.05$; c's and d's indicate significant differences at $\alpha = 0.10$). **Top**- the three treatments at twilight. Control n = 26, word- and picture-based n = 29 because we removed outliers (>3 standard deviations from the grand mean); **Bottom**- in the three treatments at night. Control n = 27, word- and picture-based n = 30. Nighttime data were transformed using the Box-Cox transformation, but values in this figure are untransformed.

ences in reaction time (braking distance) at night. These results lend empirical support to Bond and Jones' (2013) survey results reporting that drivers rate picture-based signs higher for increasing alertness. Although fewer crashes occurred in the picture-based treatment than in the word-based treatment, this difference was not significant. However, this could be the result of our relatively low sample of crashes; in a larger study with more participants, we suspect that these trends would become significant.

The reduction in mean speed at twilight from 97 km/h in the control group to 89.5 km/h in the picture-based group (7.5 km/h difference) may seem small, but previous work shows that a change of this size could greatly reduce crash probability. In a meta-analysis of crash rates before and after speed limit changes on rural roads in Europe and the United States, Finch et al. (1994) fit a model predicting that with a 1 mph (1.6 km/h) decrease in speed, there is a corresponding 5% decrease in crash rate. Taylor et al. (2002) also found that as speed increases, crash rate increases, and particularly concerning severe crashes: in their model, a 10% increase in the mean speed on a roadway predicts a 30% increase in fatal and serious crashes. Therefore, our observed differences in speed would result in tangible safety benefits for humans and wildlife.

We also found significant differences in driver behavior at twilight and at night. Average speed was lower at night, which is almost certainly due to the different speed limits at twilight and night (60 mph vs. 45 mph). This reduction in speed was accompanied by earlier braking distances and a greatly reduced number of crashes at night. The lower nighttime speed limit in Big Cypress National Preserve was put in place to reduce animal-vehicle collisions, especially collisions with Florida panthers, and our simulator data support the potential effectiveness of this measure.

Our study also re-confirmed that age plays a significant role in driver safety, with participants in age group 1 being much more likely to crash than participants in age group 3. The difference between age groups 1 and 2 was very close to being significant as well, and we believe that with a larger sample size of crashes, the difference would probably be significant. This trend highlights the need to educate younger drivers about the danger of animal-vehicle collisions.

The trends within the interaction of RADS treatment and age group on braking start distance are also worth considering. Drivers in both the youngest and oldest age groups started braking earlier in response to picture-based RADS signs. Because the animal entered the road at the same point for all participants, the distance at which participants started braking is a proxy for brake reaction time. Brake reaction time (BRT) is the amount of time that passes between the moment the stimulus appears and when the driver's foot actually reaches the pedal (Shinar 2007). Contained within BRT is perception reaction time (PRT), the amount of time that passes between when a stimulus reaches a driver and the driver initiates a response. The BRT is therefore affected by the PRT; if people are primed to expect a stimulus, their PRT (and therefore, BRT) should be reduced. The slower the PRT, the longer the stopping distance, increasing the chance of collision (Shinar 2007). Thus, it appears that for two of three age groups tested, the picture-based RADS signs did a better job of priming participants to expect an animal, and therefore may be more effective in preventing collisions.

Simulator validity

Behaviors observed in a driving simulator may not accurately reflect real-life behaviors. The validity of using driving simulators to predict real-world crash rate was reviewed by Rudin-Brown et al. (2009). Although simulators cannot perfectly recreate real-world driving conditions, Rudin-Brown et al. conclude that the use of a simulator is acceptable if it recreates conditions with enough validity to measure the behavior being investigated. Behavioral validity can be absolute—do the simulator observations exactly match those in the real world?—or relative—do simulator observations have the same direction and similar magnitude to those in the real world? (Blaauw 1982). If the goal of a simulator experiment is to measure the effect of one treatment vs. another, as is often the case in human factors research, ensuring that simulators have adequate relative validity is more important than ensuring absolute validity (Törnros 1998).

In many studies evaluating driver speed, simulations achieved relative validity but not absolute validity (Törnros 1998; Klee et al. 1999; Godley et al. 2002; Bella 2008), though absolute validity has been documented (Yan et al. 2008). It is more common for simulations to achieve absolute validity in addition to relative validity with regard to certain behaviors (reviewed in Kaptein et al. 1996), such as lateral position of the vehicle in the lane (Törnros 1998) and route choice.

Future directions

To determine whether the results of our simulator study reflect responses to the RADS in the real world, our future research will validate the simulator data using field data. We will test the relative and absolute validity of our speed results by comparing speeds recorded during the simulation experiment to field recordings of speed through the RADS zone in Big Cypress National Preserve (as in Godley et al. 2002). We predict that speeds of participants in the simulator may be higher than those observed on the road, since previous studies have shown that when the road is relatively straight, participants drive faster in simulators (Boer et al. 2000). By comparing the simulator data to real-world data, we will gain a solid foundation from which to make management recommendations about the design and effectiveness of RADS.

Conclusions

Roadside animal detection systems are a promising technology to reduce the frequency of animal-vehicle collisions, but empirical testing has been difficult because of system malfunctions in the field. We overcame these difficulties by studying the effects of a particular RADS using a driving simulator, and our data suggest that RADS can indeed produce the intended results on crash probability, driver speed, and latency to brake. With improved RADS technology, these systems could be deployed on a larger scale as a cost-effective way to improve safety for humans and wildlife.

Acknowledgements

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RESEARCH ARTICLE

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A remarkably quick habituation and high use of a rope bridge by an endangered marsupial, the western ringtail possum

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Abstract

Rope bridges are being increasingly installed worldwide to mitigate the negative impacts of roads on arboreal animals. However, monitoring of these structures is still limited and an assessment of factors influencing the crossing behaviours is lacking. We monitored the use of a rope bridge near Busselton, Western Australia by the endangered western ringtail possums (*Pseudocheirus occidentalis*) in order to identify the patterns of use and factors influencing the crossings. We installed motion sensor cameras and microchip readers on the bridge to record the crossings made by individual animals, and analysed these crossing data using generalised linear models that included factors such as days since the installation of the bridge, breeding season, wind speed, minimum temperature and moonlight. Possums started investigating the bridge even before the installation was completed, and the first complete crossing was recorded only 36 days after the installation, which is remarkably sooner than arboreal species studied in other parts of Australia. The possums crossed the bridge increasingly over 270 days of monitoring at a much higher rate than we expected (8.87 ± 0.59 complete crossings per night). Possums crossed the bridge less on windy nights and warm nights probably due to the risk of being blown away and heat stress on warmer days. Crossings also decreased slightly on brighter nights probably due to the higher risk of predation. Breeding season did not influence the crossings. *Pseudocheirus occidentalis* habituated to the bridge very quickly, and our results demonstrate that rope bridges have a potential as an effective mitigation measure against the negative impacts of roads on this species. More studies and longer monitoring, as well as investigating whether crossing results in the restoration of gene flow are then needed in order to further assess the true conservation value of these crossing structures.

Keywords
Road ecology, rope bridge, habitat fragmentation, *Pseudocheirus occidentalis*, wildlife crossing structures

Introduction

Roads can act as a barrier to movement and gene flow in wildlife populations and cause genetic isolation and fragmentation. This can results in lowered fitness and adaptability, which increases the risk of population extinction (Forman and Alexander 1998). To mitigate against these impacts, an increasing number of wildlife crossing structures are being installed worldwide because they have the potential to prevent road mortality and habitat fragmentation by providing animals with safe passages across roads (Clevenger and Wierzchowski 2006).

Arboreal species can be especially affected by roads because of their fidelity to canopies and naivety on the ground (Lancaster et al. 2011). Many rope bridges, or canopy bridges, have been installed worldwide to mitigate negative impacts on arboreal species, including several opossum, monkey, dormouse and squirrel species (Norwood 1999, Teixeira et al. 2013, Sonoda 2014). In the eastern parts of Australia, rope bridges have been built for gliders, possums, and koalas (*Phascolarctos cinereus*); however, monitoring of the use of these structures by the target species is still limited to a handful of cases (Weston et al. 2011, Goldingay et al. 2013, Soanes et al. 2013), and assessment of factors influencing the use of these structures is lacking.

In Western Australia a rope bridge was installed on Caves Road near Busselton in 2013. The bridge was targeted to provide safe crossing for the western ringtail possum (*Pseudocheirus occidentalis*), a nocturnal, folivorous, arboreal marsupial endemic to southwest Western Australia (Figure 1a). In a national action plan for Australian mammals in 2012, this species was classified as critically endangered due to a continuing dramatic decline in its numbers and range (Woinarski et al. 2014). Habitat destruction, habitat fragmentation and introduced predators such as red foxes (*Vulpes vulpes*) and feral cats (*Felis catus*) are thought to be the main causes of their decline (Department of Environment, Water, Heritage and Arts 2008, Morris et al. 2008). The Busselton region is considered to be one of few strongholds left for this species possibly because it still has a relatively high abundance of the species' main food source, the peppermint tree (*Agonis flexuosa* in the Myrtaceae Family). However, this area is also subject to rapid and large-scale developments, which threaten the persistence of the species (Jones et al. 1994a, Australian Bureau of Statistics 2014). These possums are highly sedentary and have home ranges as small as 0.31 ha in high density areas (Yokochi et al. 2015). They show a high fidelity to canopies, and Yokochi et al. (2015) found that Caves Road, a 15 m wide road was restricting their movements and home ranges. Trimming et al. (2009) also found this road to be a roadkill hotspot for this species (Figure 1b).

We monitored the use of this bridge by *P. occidentalis* and other fauna to identify the patterns of use and factors influencing the crossings. In previous studies, animals

Figure 1. Photographs of the western ringtail possum (*Pesudocheirus occidentalis*). **a** A possum at Locke Nature Reserve **b** A possum roadkill in Busselton, Western Australia.

have been observed to show reluctance towards wildlife crossing structures for a certain period of time before they habituated to them and started using them regularly (Gagnon et al. 2011). For example, possums and gliders in eastern Australia started using rope bridges after 7 to 17 months of bridge constructions, and the number of crossings increased over time until it reached asymptotes (Weston et al. 2011, Goldingay et al. 2013, Soanes et al. 2013). Therefore, we expected that *P. occidentalis* would avoid using the rope bridge for a certain period of time before it starts crossing, and that the number of crossings would increase over time and eventually reach an asymptote.

Several arboreal marsupials increase their activity ranges or change their movement patterns during the breeding season in search of mates and additional resources (Gentile et al. 1997, Broome 2001, Loretto and Vieira 2005). Other arboreal folivorous species have been observed to be less active on well lit nights with low temperatures, presumably to avoid the risk of predation and heat loss (Laurance 1990, Starr et al. 2012, Rode-Margono and Nekaris 2014). Greater wind speed also decreased the number of common brushtail possums (*Trichosurus vulpecula*) observed in open pasture (Paterson 1993). Wind speed did not influence the detection rate of *P. occidentalis* in a forest habitat (Wayne et al. 2005); however, the rope bridge in this study was completely exposed to the wind over the road, and strong wind could deter *P. occidentalis* from crossing the open bridge. Given this information, we also predicted that *P. occidentalis* would cross the bridge more during their breeding seasons and less on well lit, cold and/or windy nights.

Methods

Study area and rope bridge

In July 2013, a rope bridge was constructed across Caves Road about 9 km west of Busselton, Western Australia (33°39'32"S; 115°14'26"E) to connect peppermint trees in Locke Nature Reserve to those in a campsite across the road. Caves Road is a 15 m wide major road connecting popular tourist destinations in the region. The recorded daily traffic volume on this road was 6,000 cars in 2008, but it could vary up to 15,000 cars in the peak tourist season (Main Roads WA 2009, G. Zoetelief, Pers. Comm.). Locke Nature Reserve and its surrounding campsites are known to support the highest density of *P. occidentalis* in the Swan Coastal Plain, a region dominated by *A. flexuosa* vegetation, which is an ideal habitat for the possums (Jones et al. 1994a, Jones et al. 2007). Another possum species, the common brushtail possum, has also been observed in the nature reserve at a low density (Clarke 2011, Yokochi unpublished data).

The rope bridge was supported by an approximately 8.5 m tall wooden pole with a concrete foundation and two metal stay wires on each side of the road. The bridge was 300 mm in width and approximately 26.5 m in length. Two steel wires running between poles with nettings of marine grade ropes in between provided a flat surface for possums to cross (Figure 2). We employed the flat design over a box design because the

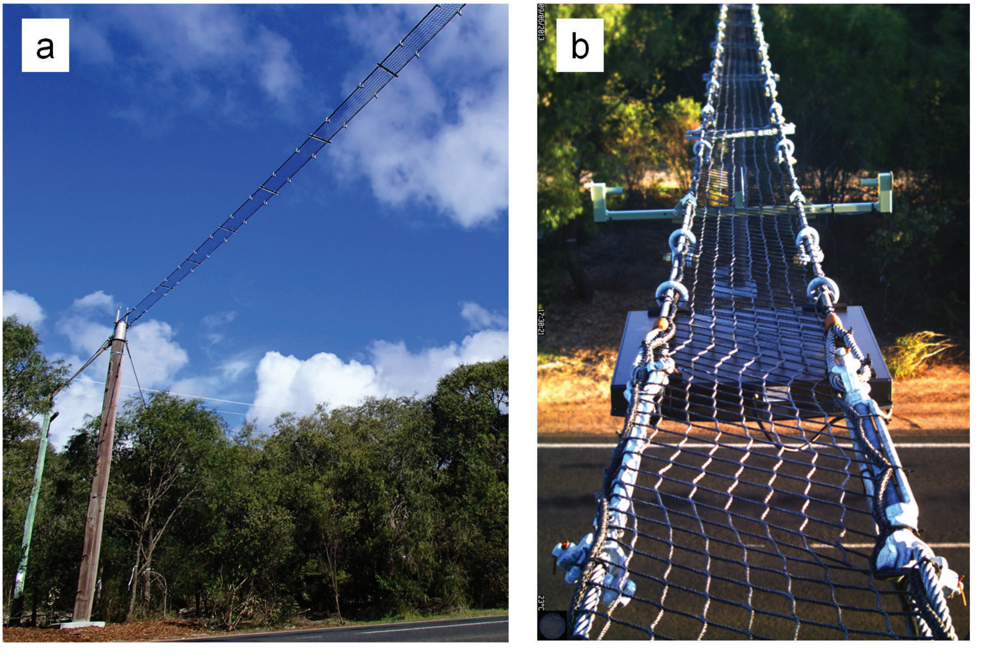

Figure 2. A rope bridge installed on Caves Road near Busselton, Western Australia. **a** Two stay wires and a rope extending from the pole of a rope bridge to nearby trees on South side of Caves Road **b** Close up of the bridge showing one of the sensors and microchip reader on the North side (taken by an infrared camera on the bridge).

box design was found to be unnecessary (Weston et al. 2011). One large rope extending from the top of each pole provided a passage between the bridge and surrounding trees, together with the metal stay wires that were in contact with nearby trees.

Monitoring

We captured 44 female and 53 male western ringtail possums within two 200 m x 200 m blocks on the North and South sides of the rope bridge site on Caves Road from March 2010 to April 2014 (Figure 3). To capture the possums, we used a specially built tranquiliser gun with darts containing a nominal dose of 12 mg/kg of Zoletil 100® (Virbac Australia, NSW Australia) following a method developed by P. de Tores and reported by Clarke (2011). A Trovan Unique ID100 Implantable Transponder (Trovan, Ltd., U.K.) was inserted subcutaneously between the shoulder blades of each captured possum.

Thirty days after the installation of the rope bridge, an infrared camera (BuckEye Cam Orion camera, BuckEye Cam Australia, Victoria), a microchip reader (LID650N / ANT612 system, Dorset Identification B.V., Aalten, Netherlands), and a pair of optical sensors were set up on each end of the bridge. When an animal moved past and blocked one of the sensors, this triggered the camera to take three consecutive photos and activated the microchip reader for a period of 30 seconds. Date and time were recorded on every photograph taken, and the microchip readers recorded the date, time and microchip code of individuals that used the bridge. Unfortunately, the microchip readers malfunctioned regularly, so we used photographic data from 270 nights of monitoring from August 2013 to May 2014 for further analyses.

A crossing was regarded "partial" if an animal was recorded on one side of the bridge only and returned back to its original side. A crossing was regarded "complete" if an animal was recorded leaving one side and then arriving on the other side within 10 minutes. We recorded the simultaneous crossing by two and three adults as two and three crossings respectively; however, a crossing by a pair of mother and young was counted as a single crossing. Species, type, and direction of the crossings were obtained from photographic data to calculate the number of complete crossings of the bridge by *P. occidentalis* on each night.

Data analyses

We used generalised linear models with a negative binomial distribution and log link function to identify the factors influencing the number of crossings per night because the crossing data were discrete and overdispersed (Byers et al. 2003). Based on our hypotheses, we constructed candidate models with variables such as days since the bridge installation, breeding season, daily minimum temperature, fraction of the moon lit, and daily maximum wind speed (Table 1). We set the breeding season as

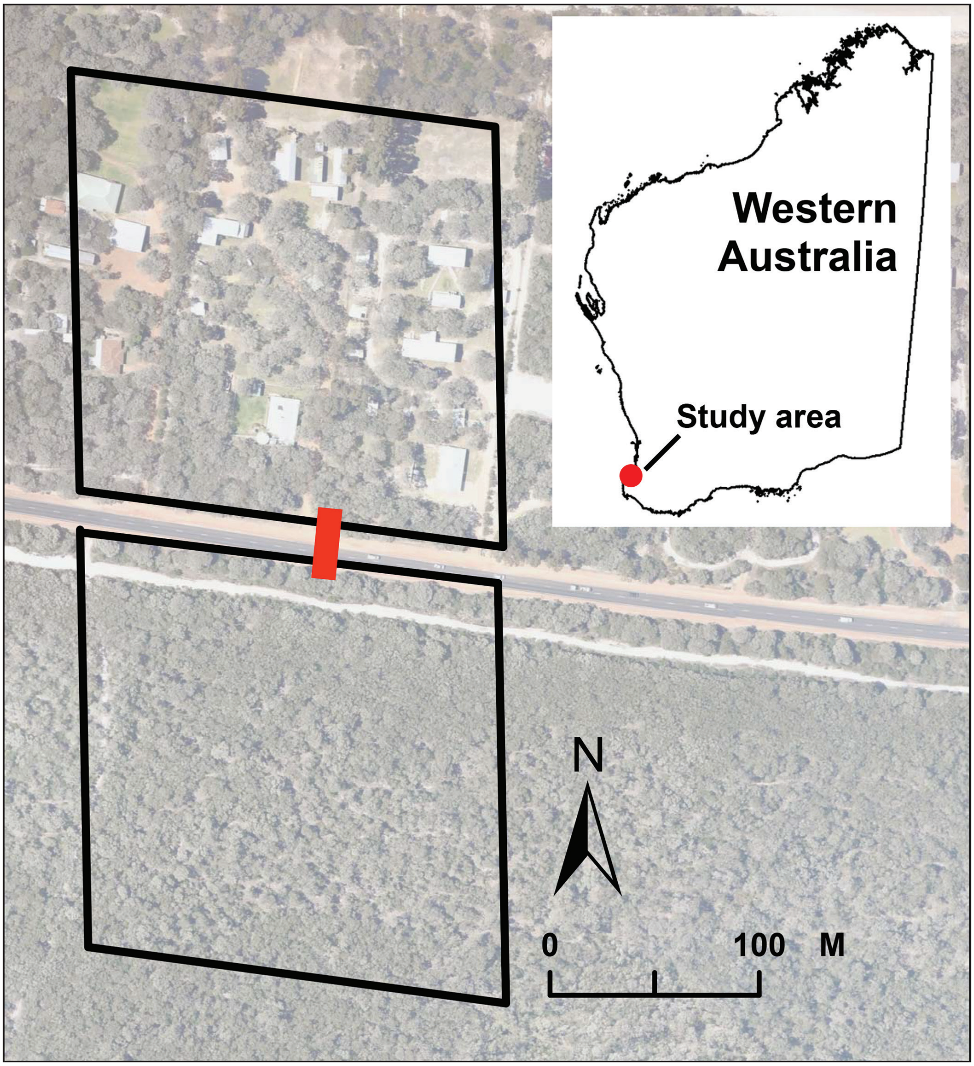

Figure 3. A map of the study area near Busselton, Western Australia. Black rectangles represent the areas where *Pseudocheirus occidentalis* were captured for tagging, and the thick red line represents the rope bridge across Caves Road.

April to July and September to November, which are the known breeding peaks for *P. occidentalis* in the Busselton region (Jones et al. 1994b). We obtained data from the Australian Bureau of Meteorology (2014) on daily minimum temperature and maximum wind speed at Busselton Regional Airport, which is approximately 15 km from the study site. Data on the fraction of the moon illuminated at 10 pm in Western Australia were obtained from The United States Navy Observatory (2014).

We ran generalized linear models using the package MASS v.7.3-35 (Ripley et al. 2014) on R v3.0.1 (R Foundation for Statistical Computing 2013, available at

Table 1. Candidate models used to analyse variables affecting the number of crossings of a rope bridge by *Pseudocheirus occidentalis*.

Variables	Hypothesis tested
Time	Crossings will increase over time.
Breeding	Crossings will increase in breeding seasons.
Min temp	Crossings will decrease on cold nights.
Moon	Crossings will decrease on well lit nights.
Wind	Crossings will decrease on windy nights.
Time + Breeding	Crossings will increase over time and in breeding seasons.
Time + Min temp	Crossings will increase over time but decrease on cold nights.
Time + Moon	Crossings will increase over time but decrease on well lit nights.
Time + Wind	Crossings will increase over time but decrease on windy nights.
Min temp * Moon	Crossings will decrease on cold nights if the moon is bright.
Null	The number of crossings varies randomly.

A generalised linear model with negative binomial regression was used to compare these candidate models. "+" denotes additive effects of variables and "*" denotes additive and interactive effects of variables.

http://R-project.org), and ranked the models based on Akaike Information Criterion (AIC) values. The model with the lowest AIC value was chosen as the best fit for the data. We considered models with ΔAIC values (the difference between the AIC value of the model and that of the highest ranking model) of less than 2 to have strong support, those with ΔAIC values between 2 and 7 to have weak support and those with ΔAIC values of greater than 7 to have no support from our data (Burnham and Anderson 2002). We also generated 95 % confidence intervals for each of the parameters to check for directionality and significant divergence from zero.

Results

Within a week of installation of the poles, an author (KY) observed two western ringtail possums on stay wires investigating the poles. This was even before the metal wires and rope nettings were installed between the poles (i.e. before the installation of the bridge was completed). Three separate partial crossings by *P. occidentalis* were recorded on 16 photos on the first night of monitoring on the North end of the bridge. The first complete crossing from North to South was recorded on the 6th night of monitoring, only 36 days after the installation of the bridge had been completed. During 270 nights of monitoring, cameras recorded 664 complete crossings from North to South and 636 complete crossings from South to North, totalling 1300 crossings. The number of complete crossings increased gradually over time (Figure 4), and *P. occidentalis* completely crossed the bridge at least three times a night for the last 100 nights of monitoring. The rate of crossings was 8.87 ± 0.59 (s.e.) complete crossings per night for the last 30 nights of monitoring. No other species, including common brushtail possums, was captured on cameras other than several

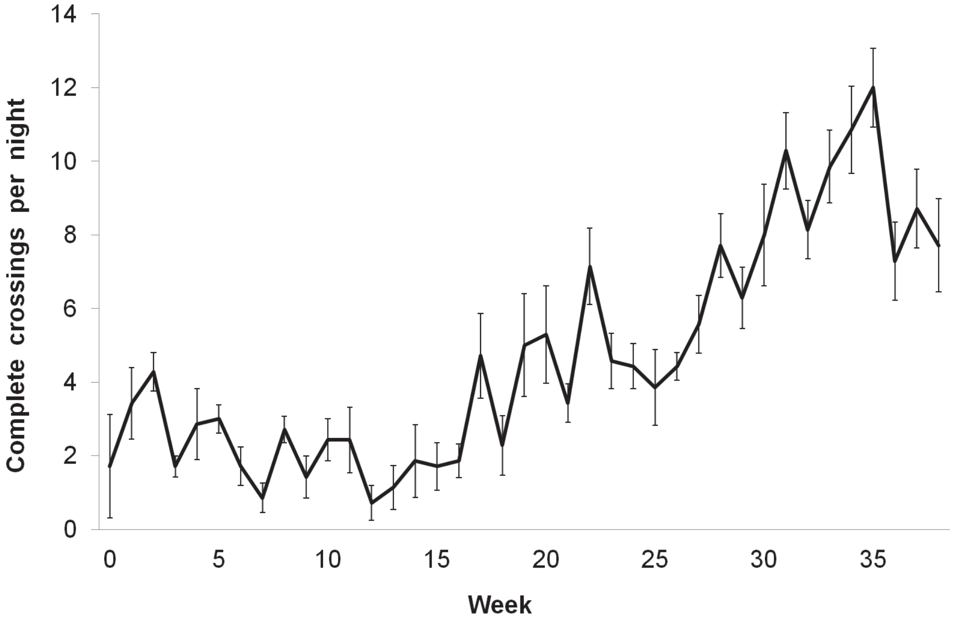

Figure 4. Weekly averages of the number of complete crossings by *Pseudocheirus occidentalis* on a rope bridge installed over Caves Road near Busselton, Western Australia. The thick line shows the weekly averages and thin vertical lines represent standard errors of the means.

birds, including Australian magpies (*Cracticus tibicen*), tawny frogmouths (*Podargus strigoides*), common bronzewings (*Phaps chalcoptera*), silvereyes (*Zosterops lateralis*), and red wattlebirds (*Anthochaera carunculata*) resting on the bridge.

Microchip readers malfunctioned regularly, and not all possums using the bridge were microchipped, so only five microchipped individuals were recorded on eight nights. The North reader recorded one female partially crossing the bridge four times on one night, and the possum was also photographed on the bridge with her young on multiple occasions. After gaining independence, her young was recorded crossing the bridge on its own. Other mothers and their young as well as pairs of a male and a female were also regularly captured by the cameras while crossing the bridge together (Figure 5).

The number of crossings by *P. occidentalis* had a strong positive correlation with time since bridge installation (Table 2). At the same time, the number of crossings decreased on nights with greater maximum wind speed. The number of crossings was found to increase on colder nights although this effect was not as strong as that of wind (ΔAIC = 1.4). The fraction of the moon lit had a considerably weaker negative effect on the number of crossings compared to wind speed and minimum temperature (ΔAIC = 4.9). The correlation between the number of crossings and breeding season was even weaker (ΔAIC = 5.4) and 95 % confidence interval of its parameter estimate included zero, indicating that the breeding season did not affect the number of crossings. The interaction effect between the moonlight and minimum temperature also had no support (ΔAIC = 159.5).

Table 2. Generalised linear model analysis on number of crossings of a rope bridge by *Pseudocheirus occidentalis*.

Variables included in the model	AIC	ΔAIC	Parameter estimates
Time + Wind speed	1291.9		Time = 0.007*, Wind = -0.011*
Time + Min temp	1293.3	1.4	Time = 0.007*, Min temp = -0.027*
Time + Moon	1296.8	4.9	Time = 0.007*, Moon = -0.211*
Time + Breeding	1297.3	5.4	Time = 0.007*, Breeding = -0.141
Time	1299.1	7.2	
Wind	1430.0	138.1	
Breeding	1448.7	156.8	
Null	1448.7	156.8	
Min temp	1449.2	157.3	
Moon	1449.6	157.7	
Moon * Min temp	1451.4	159.5	

Akaike Information Criterion (AIC) values and ΔAIC values for all candidate models and parameter estimates for the models with ΔAIC < 7.0 (i.e. models with at least a weak support). An asterisk (*) next to a parameter estimate indicates that its 95 % confidence intervals excluded zero. The variable "Time" indicates the number of days since the installation of the rope bridge, "Wind" is a variable representing the daily maximum wind speed, "Min Temp" is the daily minimum temperature, and "Moon" is a variable representing the fraction of the moon lit. For the breeding season variable, the estimate is for the non-breeding season.

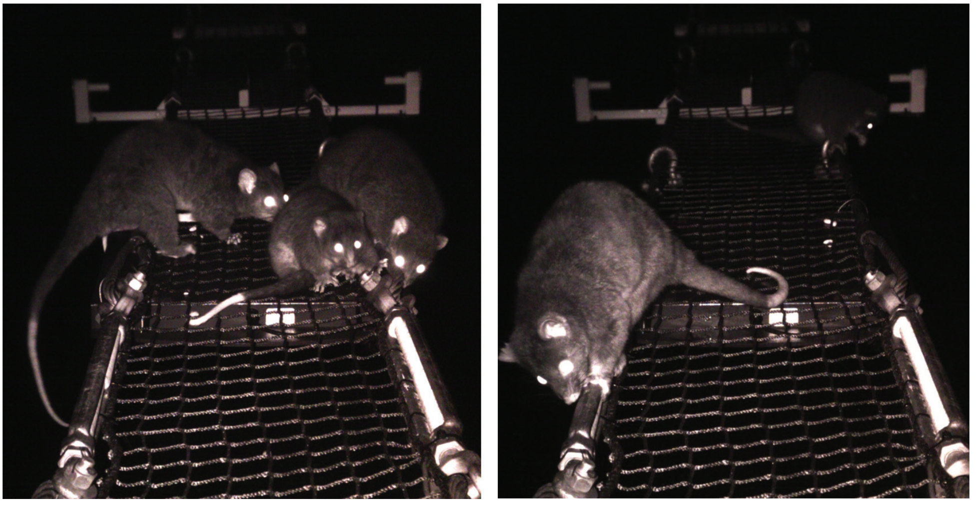

Figure 5. Photographs of mother and young *Pseudocheirus occidentalis* crossing the road using the rope bridge near Busselton, Western Australia. The left photograph is of a mother and her young with another adult possum.

Discussion

As expected, the use of the rope bridge by *P. occidentalis* increased over time; however, the possums started crossing the bridge much sooner and at much higher rates than we expected. They started investigating the bridge even before the installation was completed, and the first complete crossing was recorded only 36 days after the installation, which is remarkably shorter than seven months – the shortest time elapsed prior to other possum and glider species starting to use rope bridges in other parts of Australia (Weston et al. 2011, Goldingay et al. 2013, Soanes et al. 2013).

The rate of crossings was also considerably higher than those previously reported for other possums and gliders. Possums and gliders crossed the Pacific Highway in New South Wales using rope bridges at a rate of 0.02-0.08 crossings per night per species (Goldingay et al. 2013). On the Hume Highway in Victoria, squirrel gliders (*Petaurus norfolcensis*) used one of the rope bridges at a rate of 2.47 crossings per night after habituation (Soanes et al. 2013). In Queensland, the pooled crossing rate of three possum species was up to one crossing per 150 minutes (i.e. 4.8 crossings per 12 hours, Weston et al. 2011). The crossing rate of *P. occidentalis* recorded in this study (8.87 crossings per night) is considerably higher than these previously reported rates, and it did not reach a clear asymptote during the monitoring period. This could be due to the high density of the species in the study area and/or their particular lack of avoidance behaviour towards unfamiliar objects such as the new rope bridge (Wayne et al. 2005, Jones et al. 2007, Clarke 2011, Yokochi, K. personal observation). Moreover, possum species studied by Weston et al. (2011) lived in rainforests that generally have greater canopy cover than our study site, and their fidelity to a dense canopy may have made these possums more reluctant to cross exposed bridges. Uneven numbers of crossings by *P. occidentalis* in different directions suggest that some individuals crossed the bridge and remained on the other side. Use of the bridge by two generations of possums is also an encouraging sign that it will be used over generations and that it will be able to help increase gene flow across the road. This suggests that *P. occidentalis* can learn to use this type of wildlife crossing structure very quickly, and shows that rope bridges have the potential to be a very effective mitigation measure against the negative impacts of roads on this critically endangered species.

The number of bridge crossings decreased on windy nights, as expected. Being exposed to strong wind on the bridge may have discouraged possums from crossing due to the higher risk of being blown away. A higher risk of heat loss could be another reason for possums to cross the bridge less on windy nights (McCafferty et al. 2011); however, this is unlikely to be the case given that the number of crossings actually increased on colder nights, contrary to our expectation. It seems that heat loss is not as big problem for *P. occidentalis* as for other arboreal mammals studied by Laurance (1990), Starr et al. (2012), and Rode-Margono and Nekaris (2014). These researchers studied species in tropical regions such as Northern Queensland, Cambodia, and West Java, and their study species might have been more susceptible or less adapted to cold conditions compared to *P. occidentalis*. On the other hand, the lower number of crossing by *P. occidentalis* on warmer nights may be due to their susceptibility to overheating as they are prone to overheat and known to suffer physiologically at an ambient temperature of 35 °C or above (Yin 2006). In the study area, days with higher minimum temperatures generally experienced higher maximum temperature, which might have placed the possums under heat stress. Several mammalian species have been observed to decrease their food intake and activity under heat stress in order to reduce their heat production (Terrien et al. 2011). *P. occidentalis* may employ similar behavioural coping mechanisms and thus reduce their activity, including bridge crossings, on warmer nights.

The moonlight had a weak effect on the number of crossings, and fewer crossings were recorded on brighter nights. Whether this trend is caused by possums generally reducing their activities on bright nights or possums being discouraged to cross the exposed bridge on brighter nights cannot be known from our data. Wayne et al. (2005) reported that the moon or wind had no effect on the number of possums seen by spotlighting in a forest; however, possums are likely to act differently in a completely exposed environment such as on a rope bridge compared to an environment with greater cover from predators such as the canopy in a forest. Native owl species, such as the masked owl (*Tyto novaehollandiae*) are thought to be present in the region (Clarke 2011), and they prey on similar sized possum species in New South Wales (Kavanagh 1996). Therefore, it is possible that *P. occidentalis* reduced their activities on the exposed rope bridge on bright nights in order to reduce the risk of predation by birds of prey.

Contrary to our expectation, the number of crossings did not increase during the breeding seasons. Home ranges of *P. occidentalis* in the same area also did not change during the breeding seasons (Yokochi et al. 2015), suggesting that *P. occidentalis* do not expand their areas of activities to search for mates or extra resources during the breeding season. A longer monitoring period would be required to assess the effect of breeding season on the crossing behaviour more thoroughly because only two breeding seasons could be monitored and the rate of crossings did not reach an asymptote in this study.

Malfunction of the microchip readers made it impossible for us to identify all individuals using the bridge; however, the data still revealed that at least five different individuals used the bridge and that these individuals were using the bridge regularly. We must be cautious when interpreting the number of crossings on this bridge because a few individuals contributed to many of the crossings. At the same time, however, this also means that those individuals incorporated the bridge into their regular movement, which yet again suggests their high adaptability to this type of structure. To identify exactly how many individuals are benefitting from the bridge, we need to improve the monitoring system or develop a more reliable way of identifying individuals.

Multiple years of monitoring of the rope bridge in this study will also be necessary to investigate the long-term seasonal and yearly changes in the use of this bridge by *P. occidentalis* as well as to identify the asymptotic rate of crossing. Gagnon et al. (2011) found that elks (*Cervus elaphus*) adapted and habituated to terrestrial crossing structures over years, and some factors, such as season, time of the day and length of monitoring, that influenced the crossing frequencies in the first year of monitoring, became insignificant after four years. Given the remarkably quick habituation shown by our study species, we may be able to identify the factors influencing their long term crossing behaviours even in less than four years.

We also need to study the use of rope bridges in other areas in order to further assess the effectiveness of these structures as a wildlife crossing structure for *P. occidentalis*. Only one bridge was installed for this study due to financial constraints, and crossing patterns and characteristics are likely to differ in other areas with different population densities, habitats, and road characteristics or even for different kinds of artificial linear structures. For instance, it took *P. occidentalis* 18 months before it was recorded on another bridge installed across a newly constructed highway in Bunbury, located only 60 km away from the study area (B. Chambers, Pers. Comm.). This is possibly due to the lower density of the species in the area, recent disturbance caused by the road construction, and the greater length of the bridge (Bencini and Chambers 2014). Although it is probably unnecessary for the rope bridge in our study because of its high crossing rate, alteration of the design would be possible for the bridge in Bunbury or future bridges if the possums do not appear to habituate to them. A design to reduce the exposure and the effects of wind and moonlight may encourage possums to start using the bridges. In another study conducted in the same study area in Busselton, we found that an artificial waterway nearby was causing greater genetic divergence among *P. occidentalis* than Caves Road (Yokochi 2015); therefore, installation and monitoring of a rope bridge across this waterway is strongly recommended given the willingness of the possums to utilise these crossing structures in this area. Crossing behaviours of *P. occidentalis* is also likely to differ in areas where more competitive arboreal species, such as common brushtail possums, exists in higher densities than at our study area because they are thought to limit the activities of *P. occidentalis* (Clarke 2011).

Using individual based analyses such as parentage testing and Bayesian cluster analysis, Sawaya et al. (2014) found that grizzly (*Ursus arctos*) and black bears (*Ursus americanus*) using terrestrial crossing structures were breeding on the other side of a highway and achieved enough gene flow to avoid genetic isolation. A similar investigation into whether the crossings of the bridge by possums result in reproduction on the other side and restore the gene flow is essential in order to assess the true conservation value of rope bridges (Corlatti et al. 2009). We also need to assess whether the rope bridge provides a safe passage for dispersing juveniles, therefore assisting the restoration of gene flow. A study focusing on this life stage needs to be conducted, as well as genetic investigations to assess the change in the level of gene flow before and after the bridge construction.

Conclusion

Roads pose negative impacts on wildlife and their impacts need to be mitigated by providing safe passages especially for threatened arboreal species. The critically endangered *P. occidentalis* habituated to a rope bridge remarkably quickly, and the bridge is now regularly used by multiple individuals at a high rate every night. This shows a high potential of rope bridges as an effective mitigation measure against the negative impacts of roads on this species. More studies and longer monitoring, as well as genetic investigations into whether crossing individuals are breeding across the road and resulting in the restoration of gene flow are needed in order to assess the true conservation value of these crossing structures.

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RESEARCH ARTICLE

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The BioREGIO Carpathians project: aims, methodology and results from the "Continuity and Connectivity" analysis

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Abstract
BioREGIO Carpathians is a transnational cooperation project, co-financed under the second call of the EU South East Europe Transnational Cooperation Programme, priority area "Protection and Improvement of the Environment". BioREGIO Carpathians run for three years (2011–2013) and is a flagship project for the Carpathian Convention (article four dealing with landscape and biological diversity), its Biodiversity Protocol and the Biodiversity Working Group. The project is built on the conservation, restoration and valorisation of the Carpathians ecological continuum to enable large herbivores and carnivores to live in coexistence with modern society. The Carpathian countries are expecting a massive pressure to modernize and extend their road infrastructures. If not considering the requirements of ecological network, this run-to-development will enhance landscape fragmentation, limit dispersal and genetic exchange of wildlife species. BioREGIO applied a multi-disciplinary approach (physical, legal and socio-economic) in order to identify the most influencing barriers regarding connectivity throughout the Carpathians. Using two ArcGIS 10.0 tools in a three-step approach and a series of site visits, the continuity and connectivity analysis identified not only physical barriers but also legal aspects and socio-economic behaviour that are influencing ecological connectivity and playing a major role to conserve wildlife population. The investigation on the ground together with local experts and stakeholders enabled the adaptation of the GIS results and the development of feasible solutions to overcome the detected barriers with recommended priorities for implementing appropriate measurements to maintain connectivity and to sustain large carnivores, herbivores and biodiversity in the Carpathians.

Keywords
Ecological connectivity, physical, socio-economic and legal barriers, umbrella species, GIS

Introduction

Continuity and Connectivity in the BioREGIO Carpathians project

Ecological corridors are "linear elements which connect the core areas and serve as migrating and dispersal routes" (Tillmann 2005). A regional ecological network can provide connectivity between spatially separated populations, countering biological processes that lead to species extinction (Beier 1995, Bennett 1998, Taylor et al. 2006). Road infrastructures are fragmenting landscape structures and are thus endangering wildlife populations by reducing the options to disperse among habitat patches (Forman et al. 2003). Fragmentation increases the risk of collisions with vehicles and limits the access to resources (Jaeger and Madriñán 2011). The above quotations, among the others, illustrate the bases on which the BioREGIO Carpathians project was built. BioREGIO was the first attempt in the Carpathian mountain range highlighting in an integrated approach the necessity for the protection of biodiversity and natural heritage beyond protected areas. The BioREGIO's charge was to cope with the new challenges of modernization: deforestation, fragmentation and habitat conversion as well as with pollution and overexploitation of resources (Kock et al. 2014). One major part in the BioREGIO Carpathians project dealt particularly with *Continuity and Connectivity.* Therein the focus was put on detecting physical, legal and socioeconomic barriers. Each of these types of barriers has an impact on hindering ecological connectivity in the Carpathians. BioREGIO Carpathians aimed to point out, where the least-cost paths for the seven selected umbrella species are located for dispersing among their most probable core areas. Least-cost modelling is one of the methods used in landscape ecology to measure ecological connectivity – by representing the landscape as an energy-cost surface, least-cost paths can be calculated that represent the route of maximum efficiency between two locations as a function of the distance travelled and the costs traversed (Douglas 1994, Adriansen et al. 2003, Etherington and Holland 2013). The selection of the species of interest was done according to literature and together with the project partners, in order to identify those:

- Being representative of the different habitats of the Carpathians environment
- Being more prone to human/wildlife conflicts
- Having a different attitude towards human society
- Being ecological indicators

After several internal discussions, the species selected were: Eurasian Lynx (*Lynx lynx* L.), Brown Bear (*Ursus arctos* L.), Grey Wolf (*Canis lupus* L.), Eurasian Otter (*Lutra lutra* L.), Western Capercaillie (*Tetrao Urogallus* L.), Chamois (*Rupicapra rupicapra* L.), European Hare (*Lepus europaeus* P.).

The target of that *working approach* was to focus on the following research questions: How are the most suitable landscape patches for the umbrella species spatially distributed across the Carpathians? Are there chances for the most mobile species to reach other suitable patches? In addition, if yes, which paths are most likely appropriate to them? Are there social, legal or physical barriers in the identified routes? Are they surmountable? To reply efficiently to these questions it was fundamental to:

1. Select a Habitat Suitability Model and to define the parameters for detecting the general ecological connectivity in the Carpathians,
2. Assess the connectivity via the visualization of core areas, least-cost paths and potential barriers,
3. Perform site visits in specific locations to check the supposed barriers,
4. Perform interviews to partners and local experts,
5. Develop a web-GIS application for the visualization of the Carpathians' ecological network
6. Elaborate specific recommendations to overcome the identified barriers

Due to the necessity of a bilateral nature management in several locations in the Carpathians, common management measures and harmonized strategies were developed in transboundary ecosystems, where adjacent habitat types and nature values have to be preserved under different legal, social and economic circumstances. The following national and nature parks, located at national boarders and facing transboundary challenges were selected as pilot regions in BioREGIO Carpathians (Fig. 1):
Duna-Ipoly National Park: Ipoly-valley (HU) / Poiplie Ramsar site (SK)
Iron Gates Nature Park (RO) / Djerdap National Park (RS)
Maramures Mountains Nature Park (RO) / Carpathian Biosphere Reserve (UA).

Methods and Data

Habitat suitability and linkage model

After a comprehensive literature review (i.e, Adriansen et al. 2003, Ardeleanu and Mirea 2009, Beier 1995, Breitenmoser and Haller 1993, Forman et al. 2003, Majka et al. 2007, Salvatori 2004) on GIS wildlife habitat modelling, we identified the appropriate GIS tools for our purposes. Among the available GIS habitat suitability models, we developed a combined GIS approach using the ArcGIS 10.0 tools CorridorDesign (Majka et al. 2007) and Linkage Mapper (http://code.google.com/p/linkage-mapper/). These tools are free of charge, relatively easy to apply, adaptable to specific situations and do not require the collection of empirical data on wildlife presence. CorridorDesign is applied to create a general suitability map for the ecological requirements of certain species in particular areas. Those maps indicate for each umbrella species at pixel-level the percentage of their affinity towards a set of habitat factors (e.g. land

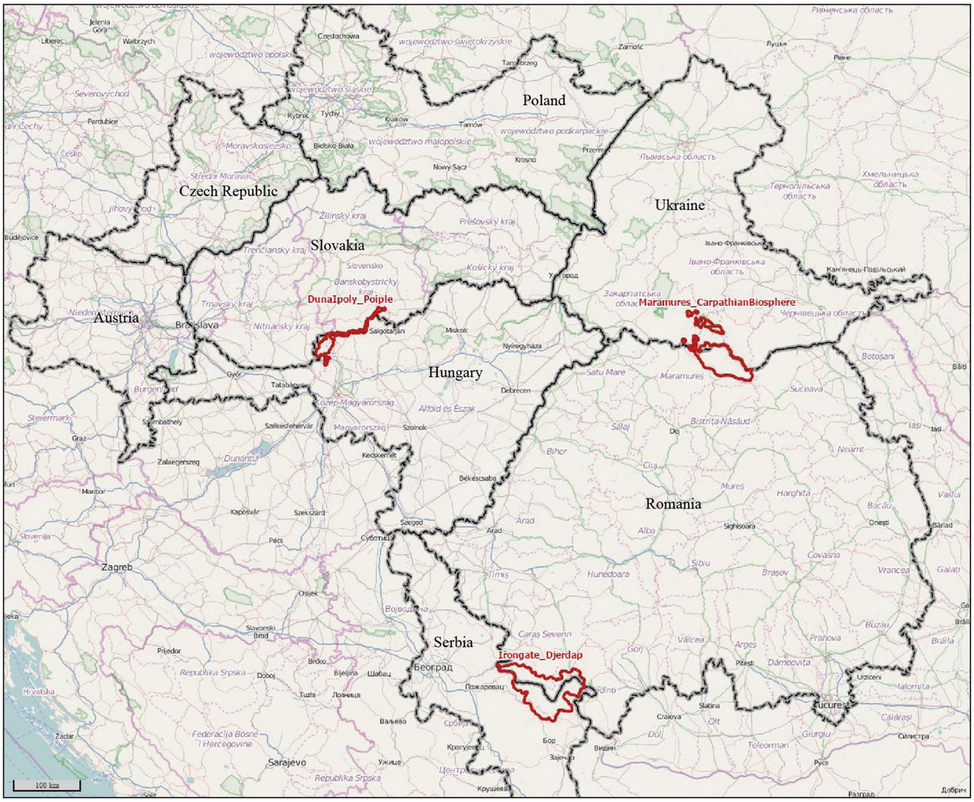

Figure 1. Web-GIS screenshot showing the BioREGIO Study site and Pilot Areas. The red polygons represent the selected national/nature parks (from http://webgis.eurac.edu/bioregio/).

cover, elevation, topography) describing the ecological framework conditions. The percentage suitability values for each species-specific factor and classes, considered as homogeneous for the whole BioREGIO study site, were taken from the literature and decided with partners and local experts. Within each habitat factor, several classes are differentiated. Every class is valuated according to its suitability towards the dispersal habit of the umbrella species For example the factor "land cover" considers classes like grassland, forest and urban areas, or the factor "topographic position" consist of classes like ridge top, canyon bottom, & steep slope.

Factors' classes and weights were combined through a geometric mean. With this approach, a pixel remains 0 if only one of the category is 0. Each pixel can then be assigned to a certain suitability class:

Suitability: 50–100% = Optimal habitat
Suitability: 25–50% = Sub-optimal habitat
Suitability: 0–25% = Occasional habitat
Suitability: 0 = Avoided, barrier

Table 1. Shows the selected factors and weights for each considered species used in our study. The weight of each factor reflects the relative importance it has for a certain species regarding its distribution and potential barrier.

Species	Land Cover	Elevation	Topographic position	Distance to Roads	Distance to urban
Lynx	40	10	20	15	15
Brown Bear	30	10	30	20	20
Grey Wolf	30	20	10	20	20
Chamois	50	15	20	5	10
Otter	40	15	20	20	5
European Hare	40	15	20	20	5
Capercaillie	40	15	20	20	5

The integration of an additional species-specific factor that concretely limits the species' dispersal (i.e., prey availability, distance from a food source, size of core area, distance of stepping stones etc.), allows the identification of the most probable core areas through the reduced extension of the general habitat suitability. For each species, the second-step factors were taken from the literature and then selected with local partners and experts (i.e., for the Lynx: Breitenmoser and Haller 1993, Kramer-Schadt et al. 2005). The pixels having a suitability values above 50% (sub-optimal and optimal habitat) were selected. In the second step, to identify the main core areas for some species, we used values taken from the literature and adapted to the Carpathians through a discussion with project partners (Table 2).

Table 2. Pixels' suitability and core areas size.

Species	Suitability	Size
Lynx	> 50%	> 10.000 ha
Brown Bear	> 75%	> 5000 ha
Grey Wolf	> 50%	> 100 ha
Chamois	> 50%	> 1000 ha

The identified supposed core areas were then used in *Linkage Mapper* to detect dispersal and connection paths requiring a minor expense of energy (*Least Cost Paths*). For a complete explanation of the model used, refer to the BioREGIO publication "Advanced tools and methodologies adopted – GIS Model Design for deriving ecological corridors" (Favilli et al. 2013, 2014).

The Least-Cost paths analysis, done through Linkage Mapper, allowed the identification of the energy spent by each species in moving from one core area to another (Cost-Weighted Distance, CWD) and of the Euclidean Distance between two core areas. The ratio CWD/Euclidean distance, calculated by Linkage Mapper, can be a clue to identify the more probable least-cost paths (LCPs) (McRae and Kavanagh 2012).

Identification of main social and economic barriers and challenges related to ecological connectivity

Three main methods were applied to collect and elaborate data for analysing the interaction between ecological connectivity and social as well as economic influences: a series of semi-structured interviews, an online questionnaire for partners and site visits at those bottleneck areas (hot-spots), the applied GIS model has identified. For obtaining relevant background information, the aspects of socio-economic impacts on ecological connectivity were enlightened in the interviews from the research view and the experiences from current practices. Hence the semi-structured interviews focused on these two main groups:

- Researchers and NGOs from the Carpathians and the Alps working on the topic of ecological connectivity: the interviews (30–40 min each) were carried out during the Forum Carpaticum in Slovakia 2012. Mainly researchers from agriculture and land use planning were interviewed.
- Stakeholders at Carpathian level – mainly from park administration and local administrations: the interviews (30– 40 min each) were carried out during the CNPA (Carpathian Network of Protected Areas event) and the Mid Term Conference of BioREGIO in Slovakia 2013.

The interviews were structured differently according to the interviewee's background and were composed as follows:

- Researchers/NGOs: in the Carpathians on the topic they research/work on and ecological connectivity, regarding critical aspects about the interaction of human activities and wildlife and possible solutions.
- Stakeholder experiences of concrete conflicts: between human activities and wildlife; concerning initiatives undertaken at local protected area/administrative level; due to the level of awareness towards the topic of ecological connectivity; regarding critical issues in the interactions among stakeholders from different sectors.

In parallel with the semi-structured interviews, an online questionnaire for the BioREGIO consortium partner has been developed. In this questionnaire, partners were first provided with a list of sectors (such as agriculture, forestry, protected area management, water management, administration); they were requested to list, for each of this macro-sectors, the stakeholder they deemed more relevant for the issue of ecological connectivity (for example, for the sector protected area managers, park directors and rangers). In a following step, the partners had to evaluate on a scale from 1 (min) to 5 (max), for each identified stakeholder category, the level of three main dimensions regarding stakeholders connected to ecological connectivity: (1) awareness, (2) influence and (3) activity. "*Awareness*" highlights the degree on how well stakeholders from different sectors are informed and know about the topic of ecological connectivity. "*Influence*" refers to the extent to which these stakeholder categories have the power to foster initiatives for promoting ecological connectivity. Finally, "*Activity*" is focusing on the degree in which these stakeholder categories actively contribute to the promotion and fostering of ecological connectivity.

Site visits and stakeholder meeting

The visual interpretation of the GIS results have shown several locations as potential "hot-spots" for connectivity. Mainly roads could be detected as barriers for connectivity, but the absence of a deep knowledge on the local (national/regional) socio-economic situation and human-wildlife conflicts, did not allow us to get a clear picture of each country. Missing data, unavailability or other issues related to data property were hindering data sharing. This forced us to organize site/field visits in specific locations of the Carpathians countries. These visits were performed in five of the seven Carpathians countries (Serbia, Hungary, Slovakia, Romania and Ukraine) at specific sites. Local partners chose the exact locations of the field investigations from several possible "hot-spots" (potential barriers for ecological connectivity) the GIS analysis has detected in a visual analysis of the Carpathians, according to the local relevancy and the socio-economic influence (Fig. 2). These explorative analyses intended to check potential physical barriers for connectivity and to discuss with local partner and stakeholders additional barriers coming from hunting, tourism, socio-economic development or legal limitations concerning transboundary cooperation.

Results

Habitat Suitability Map and Least Cost Paths for umbrella species from the corridor model

The application of the GIS *Habitat Suitability Model* to all the Umbrella Species allowed us to produce seven suitability maps for the whole Carpathians range (see Figure 3 for the Lynx).

This kind of maps are based on suitability values given to ecological factors in order to obtain a probabilistic map that needs to be verified with real empirical and field data. According to these first results and as reliable empirical data at local scale are usually not available to verify the actual presence of certain species at a specific location, the organization of site visits turned out to become a practical solution. According to their ecological preferences (extension or other ecological features), the suitability map process identified the supposed core area, where the presence of a specific species is expectable with a high probability.

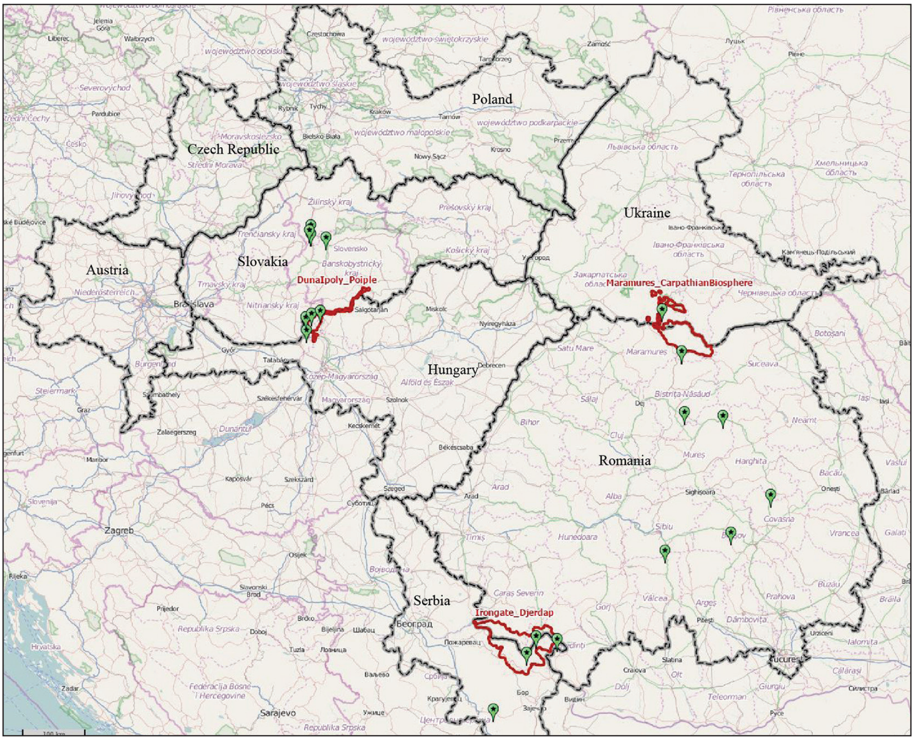

Figure 2. Web-GIS screenshot showing the selected locations (green dots) for the site visit analysis. The red polygons show the Pilot Areas (from http://webgis.eurac.edu/bioregio/).

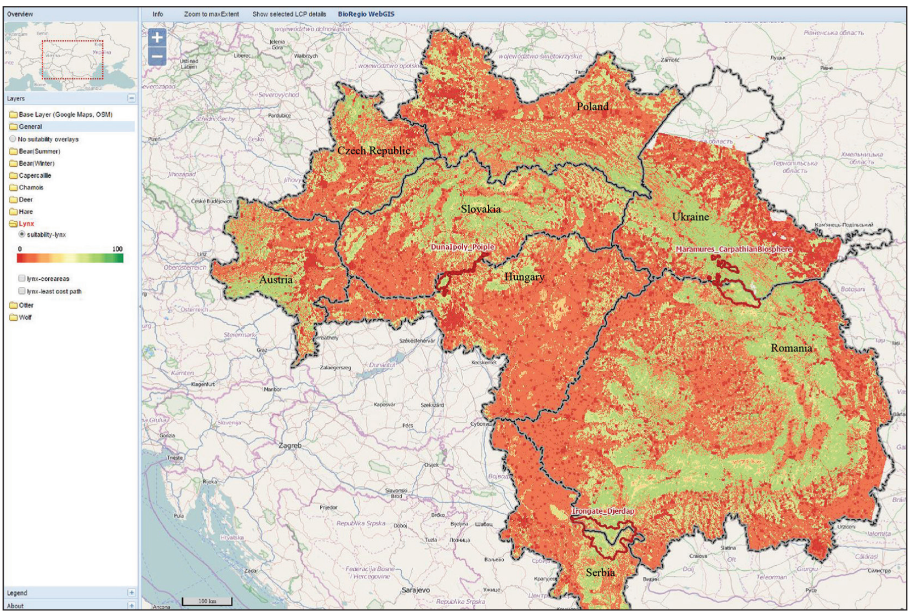

Figure 3. Web-GIS screenshot showing the Carpathians Habitat Suitability Map for the Lynx (from http://webgis.eurac.edu/bioregio/).

A case study: The Lynx in the Pilot Area Djerdap National Park (Serbia) / Iron Gate Nature Park (Romania)

Considering the overlap of the suitability maps from prey and predator species (e.g. wolf/deer; lynx/chamois etc.), it was possible to specify the potential suitability area for a certain species closer-to-reality. The subsequent application of *Linkage Mapper* to the results of *CorridorDesign* enabled the identification of the most probable paths connecting the different core areas based on the resistance of the matrix (the energy cost needed to cross a less suitable environment – see Figure 4 for the Lynx in Djerdap/Iron Gate).

The visual overlapping of the *Least Cost Paths* for each species with human infrastructures (mainly roads) enabled us, to identify the most probable physical barriers (hot spots) hindering dispersal across certain territories (Fig. 5).

Linkage Mapper therein detects all the possible *Least-Cost Paths* between all the core areas. These detected paths have to be categorized according to the CWD (cost-weighted distance): the length, the presence of barriers that increase the species' mortality risk (LCP risk), and the presence of protected areas, increasing the species' safety (LCP safe). According to the priority of these paths only those were chosen, which are more likely to be used by the selected species. In the Pilot Region Djerdap/Iron Gate, only 16 LCPs were designated to highlight the general ecological network for

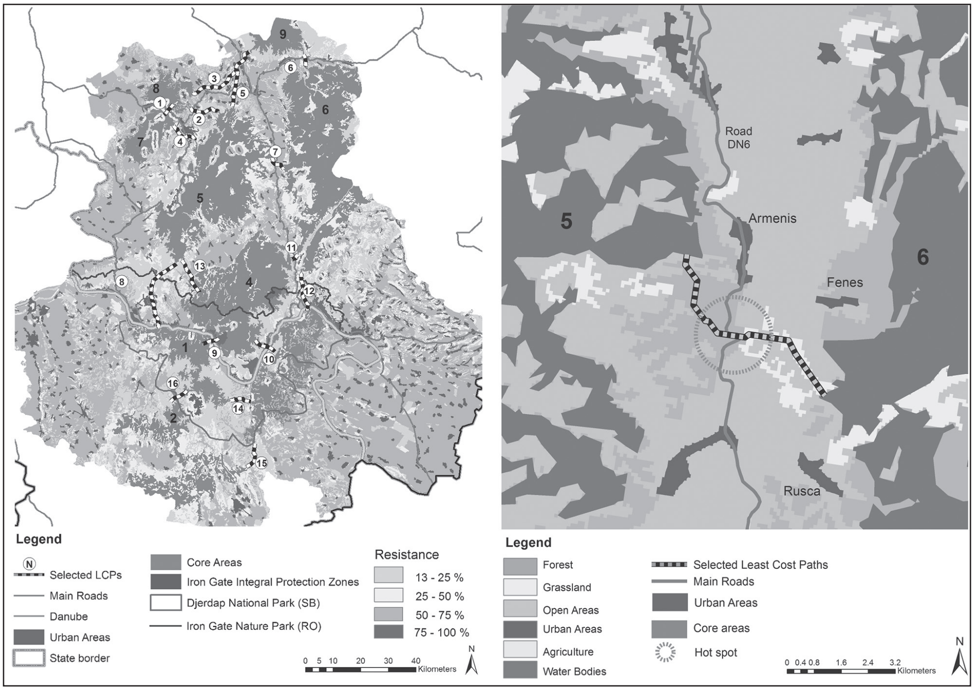

Figure 4–5. 4 Least Cost Paths for Lynx in the PA Djerdap/Iron Gate **5** Closer view of the LCP 7 and the supposed barrier.

the Lynx. Since the ratio CWD/LCP is not sufficient to identify the sites where the lynx may disperse most likely, for each species the evaluation considered the biological requirements, the impact from the different land cover types along the LCPs and the conformity to the presence of human society, to which the dispersal of these umbrella species is dependent of. The appropriateness of the LCPS for dispersal was then divided in 5 cut off categories (1 – Best; 2 – Probable usage; 3 – Possible usage; 4 – Difficult; 5 – Worst) (Table 3).

From the 16 selected LCPs, only 2 can be assigned to Category 1 because they are inside a forest in a protected area and do not meet any barrier. All the LCPs crossing the Danube have been marked with a (?) because of the uncertainties if the lynx has ever crossed it by swimming. It is more likely to assume the presence of the Danube as an insurmountable barrier, although sporadic lynx observations in this region were made and Serbian lynx populations have acquired some of the characteristics of the Balkan ones (Atanasov 1968, Paunovic et al. 2001). Nevertheless, the Danube has a seasonal changeability and for the lynx it could be passable during some winter months (Simeonovski and Zlatanova 2001, Spassov 2001). The LCPs belonging to Categories 1 to 3 seem to be usable by the lynx. The 4 and 5 ones are unlikely to be used due to the less suitable landcover classes the lynx would have to pass through and on the kind of barriers encountered.

The hereby-reported LCPs belong only to the Lynx, but the same analysis has been done for all the selected species. The selection of most likely LCPs could bring to the

Table 3. LCPs classification for the Lynx in the Pilot Area Djerdap/Iron Gate.

LCP ID	CWD (Meters)	LCP (Meters)	LCP risk	LCP safe	Land Cover*	Barrier	Ratio CWD/LCP	Usage	Category
16	34.082	1838	0	1	1	None	**18,52**	Best	**1**
14	167.728	7887	0	1	1	Agriculture	**21,26**	Best	**1**
10	87.806	3455	1	1	1-6	DN57/ Danube	**25,41**	(?)	**2**
12	83.860	3289	1	1	1-6	DN6/ DN57/ Danube	**25,49**	(?)	**2**
15	147.597	10.166	0	0	1-3	Agriculture	**14,51**	Probable	**2**
9	41.314	865	1	1	1-6	DN57/ Danube	**47,76**	(?)	**3**
13	225.601	13.340	0	0	1	Urban zone	**16,91**	Possible	**3**
1	28.443	624	1	0	1	DN58b	**45,58**	Possible	**3**
2	247.331	11.142	1	0	1-4-5	DN58	**22,19**	Possible	**3**
6	154.113	4462	1	0	1-3-4	DN68	**34,53**	Possible	**3**
8	482.063	28.736	0	2	1-6	DN57/ Danube	**16,77**	(?)	**3**
11	77.503	2997	1	0	1-5	DN6	**25,86**	Possible	**3**
7	172.290	5797	1	0	1-3-5	DN6/ Agriculture	**29,72**	Possible	**3**
4	240.811	10.315	2	0	1	DN58/ Urban zone/ Mine	**23,34**	Difficult	**4**
3	771.166	29.277	1	0	1-4-5-6	DN6/ Agriculture	**26,34**	Worst	**5**
5	679.716	23.185	2	0	1-4-5-6	DN6/ DN58b/ Agriculture	**29,31**	Worst	**5**

*Land Cover classes crossed: 1 = Forest; 2 = Grassland; 3 = Open Areas; 4 = Urban Areas; 5 = Agriculture; 6 = Water Bodies optimization of LCPs in one or more single paths for all the species. This procedure could help identifying the most important corridors for both wildlife movements and human/wildlife coexistence.

Site visits

The site visits gave the opportunity to discuss directly with local people and stakeholder the issues related to the relationship human/wildlife and the concept of ecological connectivity.

Because each country has its own history, landscape structure, laws, socio-economic environment and relationship with the local wildlife species, it was taken into consideration not only the perspective of science but also of residents, farmers and industry in order to find solutions that are practical and that may provide mutual benefits for humans and wildlife. The identified issues were then different for each country and ranged from animal – vehicle collision, building of new road infrastructures, hunting procedures and laws, forest management, intensive agriculture, trans-boundary laws, urban sprawl, and compensation of damages.

Socioeconomic barriers

The analysis on socio-economic barriers was mainly based on the preliminary semi-structured interviews and questionnaire for partners and on the site visits. Main aim was to identify the most crucial issues regarding the analyzed sectors and to propose a series of recommendations. First, this analysis provided a clear identification of the most relevant sectors connected to ecological connectivity in the Carpathians: the most relevant are protected areas, infrastructure planning, forestry, agriculture, energy, industry and the public administration at state level (ministries). Other relevant, to a lesser extent, sectors are local administrations, tourism and water management. A particularly relevant role is played, although in different measure according to the Carpathian country considered, by hunting. All these sectors have different levels of awareness and influence towards ecological connectivity. The main relevant gaps are shown in Figure 6, which represents the results of a questionnaire administered to the BioREGIO consortium. The evaluation scale for awareness and influence goes from 1 (lowest level) to 5 (highest level). It can be observed that the stakeholder group classified as the most aware (scientific community) is also the one that is considered as having the less influence. The respondents came largely from the scientific community and therefore they would possibly tend to overemphasize the perceived lack of influence. A part from that, the results nevertheless show how, often, a high influence is not associated to a high awareness.

Socio-economic barriers and possibilities take in consideration the expansion and the limitation of ecological connectivity coming not only from physical barriers. Besides, economic and social aspects have a significant impact too. This is particularly true for the Carpathian countries, which are currently experiencing quick social and economic trans-

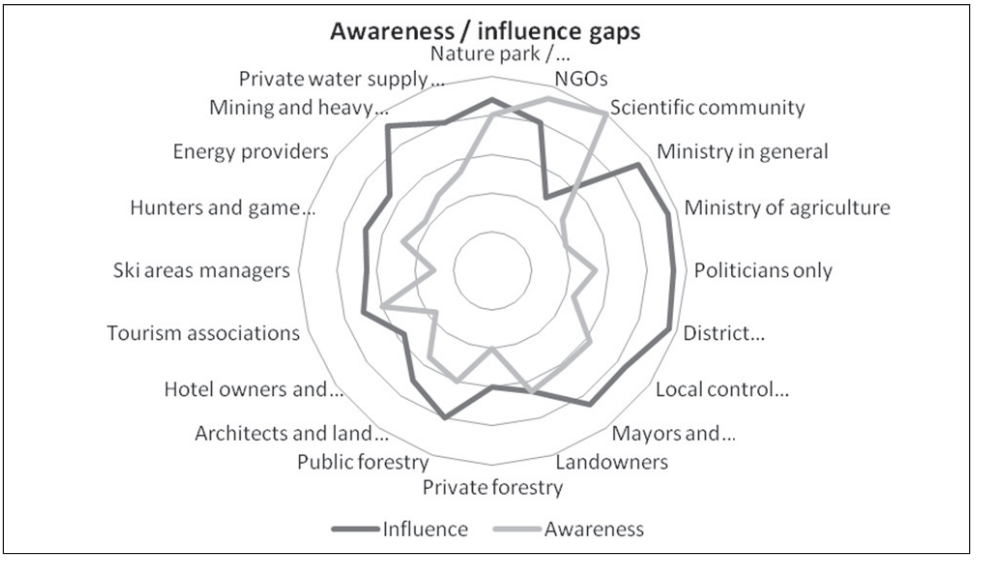

Figure 6. Awareness-influence gaps.

formation processes. Additionally, the attitude and awareness of local population towards protected areas and wildlife presence enhances significantly the effective implementation of connectivity measures. The socio-economic analysis tried to consider the impact of the different stakeholder in the different countries on the ecological connectivity

Web-GIS application

Within the framework of the BioREGIO project, a WebGIS application was designed with the attempt to spread the results of the ecological connectivity analysis and of the site visits, allowing people to know more of the structure of the Carpathians ecological network, its barriers and functionality (http://webgis.eurac.edu/bioregio/). Web-GIS applications can manage a large extent of geographical information, enabling their distribution among a large audience. The WebGIS contains both raster and vector data and it is structured into three main components: an information window; a real time maps browser with different layers containing general information concerning both the landscape and the connectivity specifically and a search engine (Fig. 7).

Discussion

Lesson learned

This suitability model developed in the framework of the BioREGIO project wanted to be a first attempt to identify the most probable areas of occurrence and dispersal

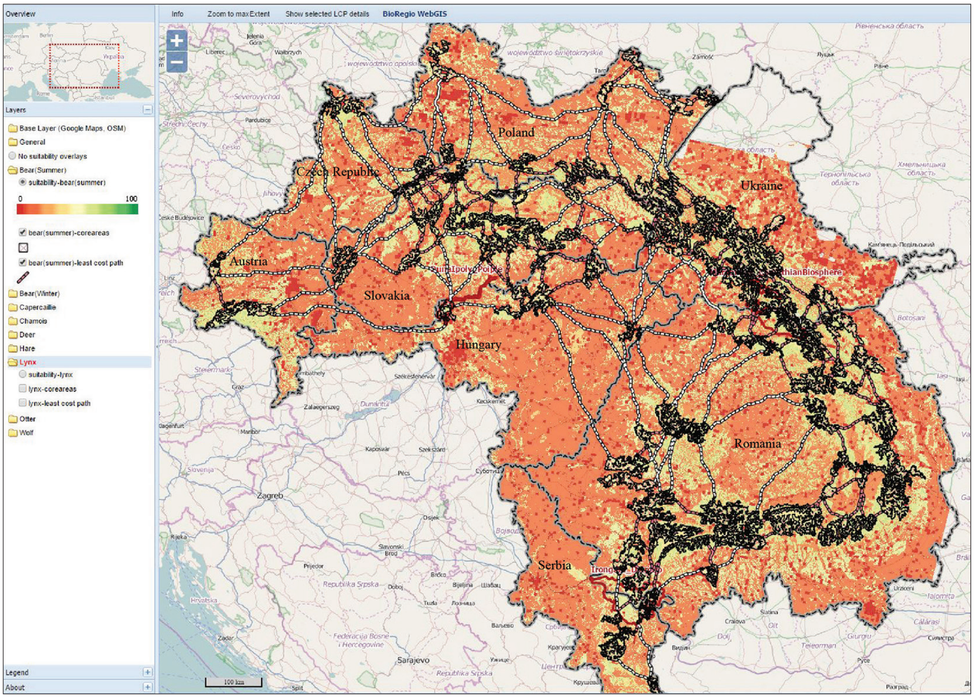

Figure 7. Web-GIS screenshot of the potential suitability map, core areas (black polygons) and least cost paths (dotted lines) for the Bear (Summer Model) in the Carpathians.

of seven Carpathians' flag species. The BioREGIO analysis started from the results obtained in previous investigations in the Carpathians such as Ardeleanu et al. 2009, Maanen et al. 2006, Salvatori 2004.

The analysis took in consideration biological, environmental and human feature in order to identify the main barriers blocking or hindering the dispersal and the socio economic situations of each Carpathians country. The obtained results do not want to be very comprehensive; the large extension of the Carpathians' arc, the different habitats and ecosystems, the socio economic and legislative aspects of the different countries require a follow-up of the investigations and actions at local scale to improve the connectivity and the human/wildlife relationship.

The GIS model tested in the BioREGIO project was a powerful tool that needed few available data to create a probabilistic map of the regional ecological connectivity. To perform a valuable analysis, it needed to receive inputs from local experts regarding the values to give to each factor's class and each factor's weight for each species. If the input data, factors, values and weights used are consistent with the actual situation, the model is able to visualize the general connectivity of the studied areas, detecting the paths that may provide safe and alternative routes. Local data on the presence and the extension of human-related infrastructures are needed to detect potential barriers to wildlife dispersal. In many cases, depending on the investigated region, this data could be obsolete and incomplete. Due to the use of the CORINE LAND COVER 2006 as a base Landcover map, some landscape features, or legal/illegal urban sprawl could not be projected in the actual way and some results could be badly interpreted or overestimated. Therefore, it was essential to perform the site visits in specific areas with the help of local stakeholder and experts to evaluate and validate the physical barriers detected through the GIS analysis and to know the local socioeconomic and legal environment. The GIS model does not want to be predictive; but the results derived from the least cost paths, the socioeconomic and legal analysis can be useful to prevent future threats to the ecological network due to the development of human infrastructures and to identify the most important corridors and the actions needed for their preservation.

Conclusions

Elaboration of recommendations

The final aim of the "Continuity and Connectivity" analysis was to produce a series of ten recommendations to overcome the barriers detected during the project lifetime. During the project lifetime, these 10 recommendations were indicated as the most relevant ones. They all refer to the initial concept to separate the barriers/possibilities influencing ecological continuum and connectivity into a physical, legal and socio-economic part. Due to the large extension of the Carpathians mountain range and to the national differences, it was intended to elaborate recommendations being applicable in all the Carpathian countries. From the practical point of view, it is almost impossible to give the derived recommendations the same priorities in each Carpathian country. It is in the nature of things that the same topic/problem is faced in different ways in different locations. That is evident due to historical reasons, the socio-economic environment, the national/local laws, the conformity of landscape, the species present causing conflicts with the human society and the personal relation of the people with local wildlife.

The ten recommendations do not want to be comprehensive. They are providing a general introduction and overview of the main barriers highlighted during the lifetime of the BioREGIO Carpathians project. The main aim of these recommendations was to look beyond the natural aspects of ecological networks and suitable areas for wildlife dispersal. Considering landscape maps is an almost straightforward strategy to define the most probable passage sites and core areas for each of the selected umbrella species. What the continuity and connectivity analysis aimed at was to define the most impacting forces influencing ecological networks, in order to prevent future fragmentation or other conflicts related to ecological connectivity.

The first five recommendations refer to physical barriers/possibilities and they concentrate on:

– New infrastructures, roads and motorways (with special focus on Deva-Lugoj Motorway, the planned D1 motorway in Slovakia and the planned M2 motorway in Hungary)

– Animal-Vehicle Collisions (highlighting the absence of mitigation structures and the driving behaviour – special focus on the road 25-1 in Djerdap National Park, Serbia)
– Hunting Procedures (the business of hunting, poaching and the impact on connectivity – special focus on feeding points in Romania)
– Forest Management (adaptation of forest management measures to promote connectivity – how to cope with economic interests and nature conservation?)
– Agriculture (Impact of intensively used agricultural fields on wildlife species – i.e., the hare)

Two recommendations refer to the legal field:
– Trans-boundary issues (cross-border natural areas, management plans, cooperation between EU and non-EU protected areas)
– Hunting laws (selective hunting, national infringements to EU legislation)

The last three recommendations refer to socio-economic topics:
– Urban sprawl and settlement expansion (the impact of (unauthorized) settlement expansion and of touristic infrastructures on the behaviour of species)
– Ecological connectivity beyond protected areas (stakeholder perception and combination of legislation and practices of sustainable integrated management)

The recommendations are providing a final assumption of topics touched during BioREGIO concerning ecological connectivity. This brief overview should enable a compact knowledge transfer, in which problems, opportunities, threats and strengths dealing with dispersal of wildlife are focused at, and in which the Carpathian particularities as well as natural assets playing herein a major role are underlined.

Recommendations are free for downloading from the project's website (http://www.bioregio-carpathians.eu/tl_files/bioregio/donwnloads_resources/Key%20Outputs%20and%20Publication/Recommendations_Connectivity_EURAC.pdf). The strategy followed took in consideration the ecological preferences of those species, their ranges of distribution and their sensitivity to human-related infrastructures. The collected information was used to develop GIS maps of habitat suitability in order to predict potentially the species' current distribution and daily/seasonal movements. Data on human activities (current/foreseen roads, settlements, hunting, forestry, agriculture and social attitude) were collected and integrated to detect the locations of possible human-wildlife conflicts and derive subsequently specific recommendations for their positive solutions. The ten recommendations are providing a general introduction and overview of the main barriers highlighted during the lifetime of the BioREGIO Carpathians project. The main aim of these recommendations was to look beyond the natural aspects of ecological networks and suitable areas for wildlife dispersal. Considering landscape maps is an almost straightforward strategy to define the most probable passage sites and core areas for each of the selected umbrella species. What the continuity and connectivity analysis aimed at was to define the most impacting forces influencing ecological networks, especially outside protected areas, in order to prevent future fragmentation or other conflicts related to ecological connectivity. Humans and wildlife share the same environment, therefore only when the factors causing conflicts are understood and solved, biodiversity together with human life could gain a higher value. Hence, it is fundamental to adapt the recommendations to the needs of the seven Carpathian countries. Based on the results of the site visits, each partner was requested to define in a questionnaire priorities concerning the importance and impact of each single recommendation in their countries and to underline their choice with a short explanation. With these essential contributions, the partners were able to derive specific approaches and recommendations that could be integrated in any legal act/guideline of a Carpathian country to sustain its ecological network and the human/wildlife coexistence.

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RESEARCH ARTICLE

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Compensation in Swedish infrastructure projects and suggestions on policy improvements

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Abstract

Environmental compensation includes a range of activities intended to counterbalance such negative impacts of development projects that remain in the environment after all preventive and corrective measures have been fully implemented. Sweden, being a member state of the European Union (EU), must implement environmental compensation under EU directives such as the Habitat Directive. However, like in other countries, implementation is not yet widespread in Sweden, and new practices and guidelines remain to be developed both nationally and at European level. This need is all the more urgent considering that the European Commission estimates that, within the EU, about 100,000 hectares of land is converted from its natural state each year.

The aim of this paper is to describe current environmental-compensation practices in Swedish road and railway projects and to discuss issues of vital importance to the development of compensation policy, such as what to compensate for, how much, and how.

A national inventory was performed, for the first time in Sweden, to identify compensation measures in road and railway projects. Data were collected from a national mailing list including 141 officials at county administrative boards (CABs), internal e-mail correspondence within the Swedish Transport Administration and databases of court decisions. The inventory focused on compensation measures ordered by virtue of the Swedish Environmental Code. In addition, two case studies were carried out to investigate the planning of compensation measures.

The results showed that CABs and courts rarely order compensation in infrastructure projects, even though this is possible under Swedish law. Between 1999 and 2012, 37 cases (i.e. permits issued) were found for which compensation was ordered. Of these cases, 76% concerned compensation for encroachments on minor habitats such as small ponds and cairns. No CAB ordered compensation for non-protected areas. Compensation ratios were never explicitly mentioned in permits, but in practice a ratio of 1:1 (often measured as area or length) was usually applied. The compensation measures typically consisted in recreating the same kind of natural asset that was affected, in a location close to the damaged area. In the two cases specially studied, the road and railway planning processes were not properly adjusted to integrate compensation issues, resulting in unnecessary bureaucracy and insufficient co-ordination between different projects, such as between the environmental-impact assessment process and the compensation process or between closely related sub-projects in the same region.

To meet the EU's goal of no net loss of biodiversity, we suggest that policy requirements should be made stricter and that incentives for voluntary compensation should be created. In line with the goals of Swedish national transport policy and the European Landscape Convention, account should be taken of social and cultural aspects, and there should be a shift from a narrow focus on individual projects to a broader planning approach, since this would allow compensation measures to be taken where they can deliver the greatest environmental benefits.

Keywords

Planning, Environmental compensation, Inventory, Infrastructure, Policy

Introduction

The idea of environmental compensation is far from new. This may not be so surprising, given that it is essentially based on the polluter-pays principle and the idea that people should make amends for their actions (Persson 2013). It may be defined as the provision of positive environmental measures to correct, balance or otherwise atone for the loss of environmental resources (Cowell 2000). Today, environmental compensation is established across the world both in legislation and in voluntary commitments by companies and local authorities (Rainey et al. 2014). However, environmental compensation is not just about making sure that developers and others abide by the polluter-pays principle. The world's access to sustainable environment is coming under ever-greater pressure and the need to take care not only of protected natural areas but also of the 'everyday landscape' is increasing both in Sweden and globally. The European Commission estimates that, within the European Union (EU), about 100,000 hectares of land is converted from its natural state each year as a result of industrial and urban development and infrastructure projects. In the Commission's opinion, somewhere between half and all of this loss should be compensated for each year (ICF GHK 2013). In Sweden, like in many other EU and other European countries such as Norway, environmental compensation has not fully come into use yet and few guidelines have been developed (Samferdseldepartementet 2013, Villarroya et al. 2014). Moreover, there is no information about how widespread the use of environmental compensation is.

Environmental compensation has been in use for longer in some countries, such as the United States and Germany, as described by McKenney (2005), Darbi et al. (2009) and Tucker et al. (2013). However, their experiences may not be fully transferable to other countries, meaning that each country and organisation must also consider its own particular context when developing guidelines and policies. In addition, there are controversial and difficult questions to be tackled, such as what environmental aspects to compensate for, what techniques to use (the available ones include e.g. re-establishment, creation and enhancement), whether to carry out compensation measures close to the damaged nature or not (on-site vs. off-site), and whether the type of compensation should reflect the damage or not (in-kind vs. out-of-kind) (Persson 2013). Another crucial issue is the problem of finding land suitable for mitigation measures (Wende et al. 2005; Rundcrantz 2007). To this should be added that infrastructure-planning and environmental-compensation processes are complex and difficult to deal with on the basis of individual perspectives from planning theory, given that they include not only strategic/rational but also incremental/pragmatic and communicative aspects (e.g. Sager 2001, Wolf and Floyd 2013, Allmendinger 2002). For this reason, the starting point of our discussion will instead be ethical principles, the European Landscape Convention and scientific research.

The aim of this paper is to describe the current state of environmental-compensation practice in Swedish road and railway projects and to discuss issues of vital importance to the development of compensation policy, such as what to compensate for, to what extent (i.e. compensation ratios), and how.

Method – inventory and case studies

No previous nationwide inventory had been performed in Sweden of the extent to which environmental compensation is used, either in general or in conjunction with road and railway projects. To form an idea of the number of compensation measures carried out for projects involving an investment by the Swedish Transport Administration since 1999, when the possibility and/or requirement to order such measures was introduced in the Swedish Environmental Code, the approach chosen involved identifying projects that had been examined on the basis of the Environmental Code and where an order to carry out compensation measures had been issued, as well as studying the conditions for compensation laid down by the authorities issuing such orders. Under the Swedish Environmental Code, the authority examining a project must order environmental compensation if Natura 2000 areas or nature reserves are affected, and it may also choose to order compensation by virtue of provisions relating to protected biotopes, the protection of species and effects on aquatic environments.

To begin with, all instances of examination by public authorities were included, i.e. those carried out by municipalities (local authorities) and county administrative boards (CABs) as well as those carried out by courts of law. A first search of legal databases (*www.zeteo.se*, *www.karnov.se*, *www.lagrummet.se* and *www.infotorgjuridik.se*) returned no court decisions; however, at a later stage of the project, two infrastructure projects were found that involved cases where courts had ordered compensation. One of the two projects involved ten cases relating to water rights, all pronounced by the Land and Environment Court of Vänersborg. The other project involved numerous decisions and appeals; it turned out to be too complex to be fully included in the inventory. The results reported below are therefore limited to decisions made by CABs and municipalities.

The inventory was carried out by means of two questionnaire surveys. One was sent out on 16 November 2012 as an internal mailing within the Swedish Transport Administration. The other was sent on 19 November 2012 to all CABs through a mailing list for officials handling nature-conservation cases; it included 141 addresses. Telephone calls were made in January and February 2013 to those CABs that had not yet responded. Here it should be noted that this approach did not necessarily capture all cases of environmental compensation, since compensation measures do not have to follow from formal examination under the Environmental Code but may also be undertaken on a voluntary basis. However, voluntary compensation is probably not that usual; only six such cases were identified through interviews and other methods.

Two cases, one from Järfälla near Stockholm and one from Umeå, were studied more closely. In both of these cases, concrete work on the environmental impact assessment (EIA), environmental compensation, etc., started around 2010, but this was of course preceded by a lengthy preparatory phase. The criteria used to select cases were that they should represent both municipal and CAB decisions and both road and railway projects, and that they should be relatively complex and involve several different aspects of compensation. Preferably, some time should also have passed so that it would be possible to study the entire planning and implementation process. The two cases selected were the only ones that fully met the criteria. We could also have chosen several smaller, more specific cases in specific counties, using a 'county' criterion as the basis for selection, but we decided to restrict our study to the two above-mentioned cases because they complement each other very well. All documents relating to the compensation issues in these two cases that we were able to find were studied closely. The remaining documentation (such as the EIA and the work schedule) was studied in a more cursory manner. Project leaders were interviewed over the telephone and given complete freedom to describe their version of the course of events. Supplementary and follow-up questions where we had identified points of unclarity were asked both over the telephone and by e-mail, and we also contacted other involved parties in individual cases (by telephone or e-mail).

Environmental compensation in Swedish infrastructure projects

Decisions taken by municipalities and CABs

The most remarkable finding is that the inventory yielded relatively few cases: only 37 decisions. It is difficult to quantify the total number of road and railway projects planned or implemented since 1999; however, it can be noted that the Administration allocates more than 4 billion euros each year to the building, operation and maintenance of infrastructure. The earliest case found was from 2004; since 2008, the average number of cases per year has been slightly below seven. Of all 37 cases identified, 12 related to railways, 22 to roads and 3 to both. Further, CABs accounted for 36 of the 37 decisions found while only a single one was made by a municipality.

By way of comparison, it can be noted that compensation requirements seem to be more common in Spain. Of all road and railway projects approved by Spanish national and regional authorities in 2006 and 2007, 40% (85 out of 214) included provisions regarding compensatory measures (Villarroya and Puig 2010). This percentage dropped to 22% (16 out of 72) for infrastructure projects approved between 2009 and 2011 (Villarroya 2012).

Table 1 shows the Swedish public authorities having made the decisions identified as well as the provisions of the Environmental Code referred to by them. Note that a decision (right-hand column) may refer to several provisions (the central, lighter columns). For some projects, a first decision was followed by a second, more specific one. In such cases, the second decision overrules the first one (meaning that this does not count as two decisions). By contrast, where a stretch of road was divided into several parts for which separate applications were submitted, this has been counted as one decision per application. The vast majority of decisions were associated with habitat protection (i.e. protection of small ponds and cairns), which the decision-making authority referred to in 28 out of 37 cases (76%), and with encroachment on nature

Table 1. Decisions taken by Swedish county administrative boards (CABs) and municipalities in 1999–2012 involving an order for environmental compensation. Note that a decision may refer to more than one legal provision; the right-hand column indicates the actual number of decisions.

	Legal provision relating to			**Number of decisions**
	protected areas [1)]	**protection of species** [2)]	**water use** [3)]	
Swedish county administrative boards (total number: 21)				
Blekinge	1			**1**
Gävleborg			1	**1**
Stockholm	1			**1**
Värmland	1	1		**1**
Västerbotten	1			**1**
Västernorrland	1			**1**
Västra Götaland	23	1	1	**24**
Östergötland	5	1		**6**
Municipality (total number: 290)				
Järfälla	1			**1**
Total	34[4)]	3	2	**37**

[1)] Chapter 7 of the Environmental Code (SFS 1998:808), which concerns, *inter alia*, habitat protection, Natura 2000 areas, nature reserves and the protection of shorelines.

[2)] Species Protection Ordinance (SFS 2007:845). Ordinances are regulations issued by the Government. This and the next ordinance were issued in implementation of provisions of the Environmental Code proper.

[3)] Water Use Ordinance (SFS 1998:1388).

[4)] Where 28 concerned habitat protection (i.e. protection of small ponds and cairns).

reserves and Natura 2000 areas. Of all 37 decisions, 34 (92%) involved one or both of these categories. No decision related to compensation for damage to the everyday landscape (i.e. non-protected nature). It also turned out that most CABs had not taken a single decision ordering compensation measures. Of the 21 CABs, 13 had never ordered compensation and 6 had done so in a single case only, meaning that 90% of the CABs had adopted either no such decision or only one. The CAB of Västra Götaland accounted for 24 of the 37 decisions (65%) on its own.

Design of requirements

Of the 37 decisions, 32 lay down specific requirements to be met by the compensation measures while 2 of them state that the measures are to be designed later and 3 of them provide for compensation in the form of an amount of money being allocated to a fund. Further, the inventory showed that the compensation measures tended to resemble the damage done and that the prevailing view is that the measures should be on a par with the intervention, i.e. that the compensation ratio should be 1:1. A typical example is a habitat-protection case in eastern Sweden (Lilla Edet on route E45) where an encroachment entailing the loss of 152 m of stone walls and 176 m of open ditches was compensated for through the addition of 150 m of new stone walls and 150 m^2 of new wetland (Länsstyrelsen Västra Götalands län 2011).

Planning perspectives

Two cases

Two cases were studied in greater detail: (1) Röbäck/Röbäcksdalen, a project to build a new bypass on route E12 west of Umeå in relation to which the CAB ordered compensation measures; and (2) Järfälla, a project involving the expansion from two to four tracks of the railway stretch between Barkarby and Kallhäll in conjunction with the Mälarbanan line between Stockholm and Örebro, where the compensation order was issued by the municipality. The purpose of these two case studies was not only to illustrate the practicalities of environmental-compensation cases, but also to highlight a few concrete examples of implementation problems and opportunities for improvement from a planning perspective.

The Röbäck case is a rather traditional habitat-protection case, but a large and complex one. Its outcome was that an encroachment on a Natura 2000 area was compensated for through the establishment of a new nature area elsewhere. It was not possible to do this within the actual Natura 2000 area because of difficulties relating to land ownership, as revealed by the interviews – a large number of private land owners would have been involved, the situation regarding financial compensation and management issues was unclear, and the Swedish Transport Administration lacks the power to expropriate land for compensation measures. Therefore it was decided to use land owned by the municipality instead. The Järfälla case, by contrast, represents a broader problem complex because the expansion of the railway will entail both an encroachment on a nature reserve and impacts on the natural and cultural environment as well as on opportunities for outdoor and recreational activities. The Swedish Transport Administration was therefore ordered to compensate for the encroachment and impacts involved, for example by moving cultural remains and by building new bridges for better access to the recreation area by foot or bicycle as well as eco-passages. The compensation process in the Järfälla case has proceeded smoothly and the various interested parties have co-operated well. Measures have begun to be planned and implemented, and issues of maintenance are being discussed.

General problems: transparency and co-ordination

One general conclusion drawn from the case studies was that both projects were highly dependent on the skill of the parties involved and on their will to carry through the compensation measures in the best possible way. Rundcrantz (2007) draws similar conclusions from her case studies. However, an environmental expert from the Transport Administration noted that it was difficult even for those involved to grasp the whole process, and that the actual outcome was finally determined by a 'tug-of-war' between different actors as well as by legal and practical issues (Persson et al. 2014). Both projects seem likely to attain good final results, but it is very difficult for external reviewers to form a clear picture of the entire planning process based on the official documents.

A further conclusion specific to the Röbäck case concerns the fact that, in recent years, the region around Umeå – especially the many bird-protection areas to be found there – has seen the implementation of a large number of major road and railway projects giving rise to orders for environmental compensation. However, in the Röbäck case it was difficult to gain access to the various formal documents that circulated in relation to each individual part of the road project, not to mention experiences made by the various involved parties that had not actually been written down anywhere. One might think that by now some form of regional co-ordination of the environmental issues concerned would have evolved. This should have been able to bring about not only a more coherent approach both to the assessments and to the interaction between them, but also a package of environmental measures yielding maximum nature-protection benefit. That would have greatly enhanced both efficiency and transparency.

As regards the Järfälla case, it turned out that sub-projects that should in theory be capable of co-ordination, such as the EIA for the work schedule and the compensation case, were in fact carried out separately and in parallel, with no systematic cross-referencing. Indeed, sometimes the same individuals were involved in preparing parallel information for different documents. As a result, those documents may be very similar in part while there are also clear differences between them; in the absence of an explanation, this may be confusing to external reviewers (indeed, it was not clear to us until we had the opportunity to ask some follow-up questions in a second interview). In all likelihood, each individual sub-project has its own planning, budget, staff, etc., meaning that the work is governed by a project-based logic rather than being part of a coherent process where the site and the problems are in focus. From the perspectives of transparency and efficiency, it should be possible to achieve improvements through a clearer formal description of how compensation cases are to be incorporated in the overall road- or railway-planning process, as previously discussed by e.g., Rundcrantz (2007). It might also be a good investment to ensure that there is a regional project co-ordinator who knows the region well and is aware of all ongoing road and railway projects, in order to avoid duplication of work, further enhance transparency and increase efficiency.

Discussion – policy implications

Why compensate?

Before discussing the policy implications of environmental compensation, it might be a good idea first to ask why anyone should undertake compensatory efforts in the first place. A list of arguments for and against compensation can be found in Persson (2011). These arguments are important to keep in mind when advocating the use of voluntary compensation. The principal arguments in favour are the following:

- There is an ethical responsibility to preserve common (i.e. public, in the sense that they are enjoyed by many people) goods such as nature. This is in line with the idea of ecosystem services: nature provides humans with benefits, and destroying the capacity of nature to do so will decrease the quality of life of humans (Cuperus et al. 2001, BBOP 2009).
- People should make amends for their actions. This argument rests on the polluter-pays principle (European Commission 2001, Beder 2006).
- The overall stock of environmental assets should not be reduced. This is also the rationale behind the principle of the mitigation hierarchy. This hierarchy consists of four (sometimes counted as three) consecutive steps: first, avoid causing impacts; second, minimise any impacts that cannot be avoided; third, restore any impacts that cannot be minimised; and fourth, compensate for or offset any remaining damage that could not be avoided, minimised or restored (see, e.g., BBOP 2009). Skipping any one of these steps may undermine environmental sustainability.

To this should be added that there is reason to believe that a smart use of environmental compensation will shorten planning processes and reduce conflicts (Persson 2006, Samferdseldepartementet 2013). Not unexpectedly, surveys on attitudes have shown there to be strong support for environmental compensation, even though they also identified some concern that the system may be abused by authorities and developers (Rundcrantz 2005, Persson 2013). In other words, there are both factual arguments and popular support in favour of the claim that the Swedish Transport Administration and corresponding authorities in other countries should pursue the issue of environmental compensation.

One way of increasing the use of environmental compensation is to improve the relevant legislation. However, considering the arguments above, it may well be that an even more important action to take in order to increase its use is to draw attention to all of its advantages so as to promote voluntary compensation.

What to compensate for?

The inventory showed that compensation had mainly been undertaken by reference to habitat and species protection, and also that no compensation order had been issued in relation to encroachment on the everyday landscape, i.e. unprotected areas, whose importance has been increasingly stressed by numerous authors (e.g., Rundcrantz 2007, ICF GHK 2013). This focus on biodiversity and on specific objects (such as wetlands and cairns) is a result of the approach presently taken in the Swedish Environmental Code, which focuses, as mentioned above, on habitat and species protection but ignores social, health and cultural aspects. This means that a number of environmental assets and ecosystem services are not taken into account. Examples of neglected issues include:

- the impact on cultural environments or characteristic features of the landscape;
- the risk of ill-health (caused, for example, by noise and air pollution);
- opportunities for recreation;
- encroachments that affect farming, forestry, fisheries or other land-dependent economic activities, such as by reducing the amount of arable land.

The legislators adding provisions on environmental compensation to the Environmental Code may not have included these issues, but they still underpin not only the concept of sustainable development but also initiatives such as the European Landscape Convention. It should also be noted that an emphasis on habitat and species protection similar to the one in Swedish legislation exists in countries such as Norway and Germany, whereas aesthetical values are given more weight in the corresponding British legislation (Thompson et al. 1997, Samferdseldepartementet 2013). Hence, even though there is a general focus on biodiversity, it is up to each country to decide what will be compensated for. When it comes to voluntary compensation, obviously, there is even greater freedom of choice.

How extensive should compensation measures be?

The inventory showed that the level of compensation ordered was on a par with the extent of the intervention – in practice, the compensation ratio was thus 1:1. There is a Swedish legal precedent in the form of a decision by the Land and Environment Court of Appeal (Miljööverdomstolen 2002) from which it follows that authorities granting permits may not demand too much: the amount of compensation must be reasonable for both sides, i.e. both for the developer and for the public interest. What, then, is 'reasonable' in this context? Below is a list of good reasons for compensating by more than 1:1.

- *Time*. The economic way of looking at the allocation of benefits and costs is based to a large extent on discounting. This means that a compensation measure is worth more today than in ten or twenty years – not because, say, an actual wetland today could not have the same functions as an identical future wetland, but because the services provided by the wetland can be used immediately (Cole 2010).
- *Uncertainty*. Whenever a compensation measure is taken, there is some uncertainty as regards how the relevant functions will develop and work in the future (Maron et al. 2012). Even if the best available knowledge is used, there are many potential problems, ranging from the specification, implementation and maintenance of the work to the possible failure of nature to abide by the scientific models believed to be valid. Such shortcomings in models may relate, for example, to how a certain species goes about establishing itself or to how a water system reduces the amount of nutrients.
- *Quality*. Many US states have guidelines for how to weight different technical solutions when comparing them. For example, restoration is seen as more valuable than the creation of new environments, and the creation of new environments is seen as more valuable than the granting of protection for an area (National Research Council 2001).
- *Actors involved*. When the reasonableness of measures is discussed from the perspective of the different actors involved, issues specially focused on are whether the assessment of reasonableness differs between actors, how they perceive situations differently and how specific mitigation measures would work in each of their contexts (e.g. in out-of-kind and off-site compensation). This is especially important when it comes to compensating for social aspects (Persson 2006).

When ecological criteria are applied, high compensation ratios may be needed to guarantee a fair exchange (Maron et al. 2012). It should be noted that such ratios may be perceived as 'over-compensation' by certain actors, since other criteria (such as hydrological models) will typically yield lower ratios. Also, high compensation ratios may be needed to compensate for damage to recreational and other social areas. Further, it should be kept in mind that environmental disputes typically involve several perspectives and interests besides the matter of the correctness of scientific models (Söderbaum 2000, Persson 2006). Ideally, a ratio that is high enough to satisfy all environmental aspects that need to be compensated for should be chosen.

How and where to compensate?

It is clear from the inventory that Swedish decision-making authorities have consistently proposed compensation measures of the same type as the intervention and usually also located close to the intervention (i.e. 'in-kind' and 'on-site' compensation). This was also the case in Järfälla (however, an exception could be seen in the Röbäck case, where the mitigation measures were out-of-kind and located off-site). This is in line with a principle which is advocated in the guidelines of the Business and Biodiversity Offset Programme (BBOP 2009) and which has been very influential not only within the EU (Tucker et al. 2013) but also in non-EU countries such as Norway (Samferdseldepartementet 2013). By contrast, others have argued that it is important to take a flexible view and adjust the compensation measures to suit each case, because each case is unique – or, as Cuperus et al. (1999:46) put it, '[g]eneral standards and guidelines for choosing between "on-site/off-site" and "in-kind/out-of-kind" compensation cannot be given, as these depend on the availability of suitable compensation sites and must therefore be determined on a case-by-case basis'. Paradoxically, alongside the BBOP's focus on on-site compensation there is a worldwide trend for compensation to be provided off-site because of problems finding places for the environmental measures. This trend is likely to continue in Europe and elsewhere, given that mechanisms such as 'habitat banking' are also promoted by the European Commission (Conway et al. 2013) and national governments such as the ones in Germany (Wende et al. 2005) and the United States (Race and Fonseca 1996, Federal Register 2000, Stein et al. 2000, U.S. Army Corps of Engineers 2010).

The core of this issue can be summed up as follows: on-site and in-kind compensation is a way to make a contribution close to the site of the damage in order to safeguard local assets (such as ecosystem services), while off-site and out-of-kind compensation may be a necessary option for the promotion of assets of a more general type deemed to be more important or in cases where it is not possible to find an on-site and in-kind option, i.e. in the context of a strategy governed by objectives (Villarroya et al. 2014). The use of the kind of overall landscape perspective characteristic of the latter approach leads to greater flexibility and makes it easier to find locations for compensation measures. In addition, landscape-scale planning allows a more effective implementation of the mitigation hierarchy and prevents overlooking cumulative impacts (Kiesecker et al. 2010). It is true that objections have been made to off-site and out-of-kind compensation based on the hypothesis that it may be easier to gain the acceptance of the general public for on-site and in-kind compensation (ten Kate et al. 2004). However, no studies confirming this have been found and a study of preferences as regards environmental compensation actually showed the opposite hypothesis to be closer to the truth (Persson 2013).

Conclusions

A national inventory was performed, for the first time in Sweden, to identify compensation measures in road and railway projects. The results showed that CABs rarely order compensation in infrastructure projects, even though this has been possible under Swedish law since 1999. When compensation measures are ordered, they are typically intended to compensate for encroachments on minor habitats such as small ponds and cairns (76%) or for damage to nature reserves or Natura 2000 areas. This is because these aspects are the ones addressed in the Swedish Environmental Code, and so it is in relation to them that developers need to obtain permits from county administrative boards. Hence legislation plays a crucial role in determining what is compensated for. No cases were found where compensation was ordered for non-protected areas. The actual measures mainly amount to the recreation of the same kind of natural asset that was affected, with a compensation ratio of 1:1. In the two cases specially studied, the road- and railway-planning processes were not properly adjusted to integrate compensatory issues, resulting in unnecessary bureaucracy and insufficient co-ordination between different projects, such as between the EIA process and the compensation process or between closely related sub-projects in the same region.

The study shows that the use of environmental compensation is neither well developed nor widespread. To increase the use of compensation, there is a need for further policy development – but it may actually be even more important to draw attention to all of the advantages of environmental compensation in order to promote voluntary compensation and compensation directed towards non-protected areas, i.e. people's 'everyday landscape'. It may be useful to consider the following aspects in the development of new practices, both in Sweden and elsewhere:

- 'Nature' or 'the environment' should be understood broadly to encompass all relevant dimensions, including recreational opportunities, farmland, noise, climate, culture, etc.
- Compensation ratios higher than 1:1 should be used in order to take account of the perspectives of different actors and the issues of time, uncertainty and quality.
- To prevent inconsistencies when several projects are being planned in parallel, an overall landscape perspective yielding increased flexibility should be used, so that cumulative impacts can be taken into account and so that the effectiveness of mitigation and compensation measures can be enhanced. When used within the context of the mitigation hierarchy, tools such as habitat banking may assist the implementation of compensation in certain cases.
- The planning process should be transparent and well co-ordinated with other regional activities.

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REVIEW ARTICLE

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Minor rural road networks: values, challenges, and opportunities for biodiversity conservation

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Abstract

Roads corridors are a conspicuous part of most landscapes, which are gaining greater recognition for their role in nature conservation. However roads cause wildlife mortality, alter water and nutrient flows, change local microclimatic conditions, act as vectors for weeds and pest animals, and have other far-reaching effects. Not surprisingly, there is much attention from both road and conservation managers to lessen these impacts, with an emphasis on developing solutions to mitigate the barrier effects of major roads to wildlife movements. However in many anthropogenic landscapes, road corridors can also provide key habitat and connectivity for local biodiversity. In particular, where traffic volumes are low, minor roads often provide critical habitat and refuge for many native species. Knowledge of the ecology and biodiversity conservation values of minor rural road verges has been underpinned by studies in various contexts, such as sunken roads, field margins and hedgerow networks in Europe, to stock routes in Spain and Australia. Despite their different histories and management constructs, important commonalties have been highlighted in terms of their biodiversity values, and the factors which influence these values. As such, minor rural road networks can be vital in providing connected, functioning ecosystems within rural landscapes. The importance of vegetated minor rural road networks will only become more pressing with future climate change. In Australia, road management authorities are tasked with the dual roles of maintaining road transport needs (i.e. priorities for road maintenance and safety concerns), whilst maintaining the environmental values of roads. This paper reviews the biodiversity values of minor rural roads, discusses the challenges and constraints in managing these values, and describes the case of identifying historic roads as an example of enhancing conservation management of these important habitats in rural landscapes.

Keywords

Linear corridors, road ecology, roadside vegetation, stock route

Introduction

Intensive agriculture has caused irreversible damage to many ecosystems and constitutes a major threat to biodiversity (Sachs et al. 2010). Agricultural-related practices such as land clearing, intensification, inappropriate water-use, and over application of fertilizers have accelerated biodiversity losses worldwide (Conacher and Conacher 1995, Pretty et al. 2010). Such human modification and destruction of habitat has transformed once continuous ecosystems into landscapes dominated primarily by mixed pastures and food crops (Lindenmayer and Fischer 2006). Due to changing economies and policies, agricultural (and other human modified) landscapes often experience frequent and extensive land-use changes, the history of which continues to shape present-day landscape patterns and processes (Foster et al. 2003, Watson et al. 2014). This process results in extensive fragmentation of remaining habitat, creating isolated remnant habitats of varying size and quality (Saunders et al. 1991). Despite these changes, road transport corridors can provide vital refuge for species and remnant ecological communities (Bennett 1991, Votsi et al. 2012).

Roads and other infrastructure corridors are a conspicuous part of most agricultural and other human modified landscapes, and have significant impacts on adjacent ecosystems (Lugo and Gucinski 2000). For example in Europe, the impact of expanding road networks and associated urbanisation was recently described as effectively 'cutting to pieces' local ecosystems (BISE 2014), resulting in significant biodiversity losses – even in protected areas (EEA 2014). The rapid expansion of road networks into natural areas or farming systems in recent decades has become a global phenomena impacting upon production (Forman et al. 2003, Hoffman et al. 2014). The need to produce more food from dwindling farmlands means further intensification, and further pressure to upgrade existing rural road networks (Pauwels and Gulink 2000). As a result, potential positive effects of roads for biodiversity conservation may be eroding, as human populations expand.

Road networks typically consist of a system of interconnected road corridors, which are usually classified into the following categories: highways (paved), main roads (paved), minor secondary roads (heavily populated areas – usually paved; unpopulated rural areas – usually unpaved, gravel roads) and tracks (natural earth surface) (Spooner et al. 2004, FHWA 2013). Road surface conditions usually reflect the frequency of use by vehicles, where motorways and other major highways can account for up to 65% of vehicle movements (UKDT 2013), yet constitute a small proportion of the entire network length (Hambrey Consulting 2013). Although a small component of road networks, highways and other major roads constitute a significant barrier to animal movements, are a major cause of wildlife mortality, and can cause significant vehicle damage. As a result, most attention by environmental researchers has focussed on the negative effects of highways in relation to wildlife (e.g. Trombulak and Frissell 2000, Forman et al. 2003, Underhill and Angold 2000), and methods to mitigate the conflict between transportation and nature (Morelle et al. 2013).

Given the depauperate state of human modified landscapes, there is growing attention on the role of minor roads to conserve biodiversity. Although minor roads are primarily used for transportation, their biodiversity conservation values have long been recognised, and underpinned by various studies and contexts: from roadside environments in Australia and elsewhere, to field margins and hedgerows in much of Europe (e.g. Perring 1967, Foreman and Baudry 1984, Riffell and Gutzwiller 1996, Spellerberg 1998, Dover et al. 2000, Lugo and Gucinski 2000, Freemark et al. 2002, De Blois et al. 2002, Marshall and Moonen 2002, Spooner and Lunt 2004, Deckers et al. 2005, Huijser and Clevenger 2006, Noordijk 2009). Minor rural roads are the most common type of road, and frequently harbour the last vestiges of quality habitat (Bennett 1991). As a result, minor roads often provide important refuge and connectivity for many threatened species (e.g. Bennett 1991, Dennis 1992, Dover et al. 2000, van der Ree 2002, Spooner and Lunt 2004), particularly in low populated rural areas. However in more heavily populated areas such as Central Europe and the eastern states of USA, environmental change has been such that few elements of previous ecosystems remain.

Minor roads are usually narrow, possess a high edge: interior ratio, and are maintained by humans (Forman 1991). Therefore they are subject to a suite of influences from the surrounding environment (Forman and Alexander 1998, Joly et al. 2011) making them highly susceptible to invasion by exotic species (De Blois et al. 2002, Jauni and Hyvönen 2010, Schmitz et al. 2007, Kalwij et al. 2008, Šerá 2010). Various studies have shown that roadsides are often colonised by opportunistic pioneers that enjoy increased nitrogen (Cilliers and Bredenkamp 2000, Zwaenepoel et al. 2006) and available light conditions (Lavoie et al. 2007). Owing to their linear configuration and connectivity to other habitats, minor roads are often been blamed for acting as conduits for plant invasions (e.g. Gelbard and Belnap 2003, Dark 2004, Pauchard and Alaback 2004, Kalwij et al. 2008, Barbosa et al. 2010, Brisson et al. 2010). Many species can disperse over long distances throughout human modified landscapes because roads create continuous habitat over many kilometres (Christen and Matlack 2006). Minor roads can also facilitate plant invasions into adjoining ecosystems, particularly when the road corridor intersects areas of intact habitat (Tyser and Worley 1992, Flory and Clay 2006, Vilà and Ibáñez 2011).

The extent to which minor roads provide biodiversity conservation benefits for species, as opposed to promoting alien flora and fauna, is largely dependent past land-use history and ongoing disturbances experienced in roadsides (Spooner et al. 2004, Hansen and Clevenger 2005, Barbosa et al. 2010). The maintenance of minor roads can alter soils, light levels, dust, patterns of water runoff and sedimentation, and introduce nutrients and other pollutants to adjacent roadside environments (Gelbard and Belnap 2003, Trombulak and Frissell 2000). Unlike natural disturbances such as fire, disturbances in roadsides are often repeated more frequently and with far greater intensity than most natural events (Frenkel 1970, Hobbs 1987, Spooner et al. 2004). As a result, roadsides are highly susceptible to invasion by exotic plant species (Tyser and Worley 1992, Vitalos and Karrer 2009, Jauni and Hyvönen 2010). However some studies have shown that disturbances associated with road maintenance can play an important role in the recruitment of native plants into roadsides (e.g. Spooner et al. 2004, Bognounou et al. 2009). For example, disturbance events can assist threatened species with limited dispersal mechanisms (Eriksson and Eriksson 2000) and assist grassland flora to disperse into roadsides (Tikka et al. 2001).

The growing body of literature (above) suggests that conserving biodiversity in minor rural road corridors is indeed problematic; where competing human-use values, inherent complexity of the roadside ecosystems, and limited resources are contending issues for management to deal with. However greater recognition of the need to develop road networks in a sustainable manner provides the impetus and incentive to identify new opportunities for restoration and biodiversity conservation measures to be undertaken (Dolan et al. 2006). As minor rural roads constitute a significant proportion of remnant native vegetation in many agricultural areas, conservation of roadside habitats can complement efforts to achieve more sustainable agriculture, and build greater ecosystem resilience in preparation for climate change. The aim of this paper is to discuss the management of minor roads from an environmental viewpoint, by describing the competing values, management considerations and new opportunities to explore in conserving biodiversity in rural landscapes.

Australian context – history of minor roads

Roadside environments are a ubiquitous component of the Australian landscape. Fringing a vast network of minor rural roads, remnants of native vegetation still exist alongside many Australian roads, and provide the only remaining evidence of once extensive tracts of forests, woodlands and grassland ecosystems. In conjunction with stock routes (see below), roadsides often constitute a significant proportion of native vegetation remaining in agricultural or urbanized areas, and provide important refuge for threatened or endangered species and ecosystems (Figure 1). As such, this infrastructure is one of the most extensive networks of its kind in the world. Australia is fortunate to possess this 'green' network, but how did they get there?

The history of road development in rural areas of Australia has been described by Spooner (2005) and others. Briefly, the narrow area of land which contains the road and surrounding environment is dedicated as a road 'reserve' – an area of public land set aside for transportation needs during European colonisation of Australia in the early 19th century. As landscapes were surveyed and subdivided for settlement, an extensive network of road reserves was also surveyed so all land titles could access water. Most road reserves were originally surveyed at one chain (20.12 m) width to facilitate the transport of horse and carriages - which is barely enough room for modern-day transportation needs.

To mark out the boundaries road reserves, a line of blazed trees were marked 'with a broad arrow at least 6 inches long', where a shield was marked on the tree by stripping off the bark. As road construction was a low priority in the late 1800s, most of the 'roads' formed in these land reserves were no more than a boggy collection of unformed earthen tracks. Depending on usage by transport, major roads were surveyed at much greater widths of up to 2 or 3 chains. In this way, extensive linear tracts of land were retained for transport use, and later cleared of vegetation for roads as neces-

Figure 1. A minor road in a rural area of south-eastern Australia, showing remnant Eucalyptus wood lands located within the road 'reserve'.

sary. The (indirect) legacy of these past land-use decisions is an extensive network of vegetated corridors traversing the country (Spooner 2005) (Figure 2).

Travelling Stock Routes (TSRs) are also an integral component of Australian minor rural road networks. In the 19th century, many stock routes were surveyed up to ½ or 1 mile wide, however most are now 3 chains (60.2 m) in width. It has been suggested that many TSRs originated as trails of indigenous peoples, tracks of native animals, bullock tracks of early explorers or overlanders, or as routes between early settlers homes, water-points and townships. The origins of TSRs are therefore of great historical interest, as they are a lasting imprint of people and transport patterns from long ago (Cameron and Spooner 2010, Spooner et al. 2010). Owing to their width and extent, TSRs are gaining attention for their conservation values, where many still exist as part of the minor rural road network. Similarly, the conservation importance of stock routes is gaining renewed attention elsewhere in countries such as Spain and Britain.

Management of minor rural roads – challenges

In the early 20th century, the road network in Australia amounted to no more than an ad hoc collection of minor rural roads, where travellers navigated their way through vegetated areas on poorly formed tracks. Newly formed local government (council) authorities

Figure 2. A typical road network in south-eastern Australia, showing a major highway tracking through a gridwork of vegetated road reserves, most of which are only 20.1 m wide. This region was previously vegetated with grassy woodlands, which are now mainly confined to roads and other small reserves.

were confronted with the enormous task of making this network trafficable, where road reserves that were actually being used for transport were then declared as 'open' roads. Development of road networks continued throughout much of the early 1900s, where open roads was cleared of vegetation to formally develop roads, and other road reserves eventually closed. These closed road reserves still exist as narrow vegetated corridors, providing enormous opportunities for conservation projects in rural areas (Figure 2).

Since the 1980s, there has been increasing attention toward the conservation management of rural roads and stock routes for their biodiversity and connectivity values (e.g. Bennett 1991, SANVC 1992, Hobbs 1992, Lord 1992). Bodies such as the NSW Roadside Environment Committee (REC) were formed to encourage the better management of the roadside environments, by providing training and assistance to identify the natural and cultural heritage values of roadsides. For example, most council's have now used a rapid bio-assessment methodology of some kind to assess the conservation values of each road segment (ranked as High, Medium or Low). These rankings are then used to determine appropriate management actions for each road category, as described in local roadside management plans (NSW Roadside Environment Committee 2014).

However the management of minor roads is problematic, where legislation requires councils to address a number of competing values, and manage roads accordingly

Table 1. Overview of competing values and management considerations of minor rural roads [1]

Roadside value	Description and management considerations
Connectivity	Prime function of road for humans Wildlife collisions (links to road safety) Dispersal conduit for species
Cultural heritage	Historic bridges, aqueducts, cuttings, Location for scar trees, monuments, memorials, and other built objects of historic significance, Historic road, stock routes, drove roads, driftway, sunken roads.
Ecotourism values	Route to explore sites, localities or landscape Interpretation signage and other infrastructure
Environmental	Refuge for threatened species and ecosystems Seed source for revegetation activities Weed and pest management Fire and timber management Provision of ecosystem services e.g. pollination
Infrastructure corridor	Corridor for water supply, electricity, gas and telecommunications Vegetation clearance for utilities
Recreational	Sight-seeing, horse riding, hiking, bike riding educational values
Resources	Source of firewood/ rocks/ gravel and sand Stock grazing for fodder during drought Stockpiling of materials for road management
Roadside amenity	Aesthetic values of roadside Litter management Transport parking areas
Transport and road safety	Legal requirements/ insurance, Road upgrade, construction and maintenance requirements, Soil stability management/ landslides, Water runoff, run-on control, flooding, snow and ice.

([1]) Adapted from Pauwels and Gulinck 2000, NSW Roadside Environment Committee 2014 and works by the author.

(Table 1). For example, efforts to conserve biodiversity in road reserves must be balanced with road safety and fuel reduction priorities. This is no easy task, where efforts to maintain roadsides vary enormously from one council jurisdiction area to the next. Unfortunately, many rural councils cannot afford a dedicated environment officer, and so compliance to any roadside vegetation conservation plan is often lacking. Roadside vegetation plans need to be promulgated in local by-laws for compliance to take place. A further problem is that roadside environment conservation training for workers is often lacking or non-existent, and local contractors are often over-looked in the training process. Ongoing training and monitoring is critical to ensure that workers know exactly where high quality/ conservation value sections are, so as to avoid damage by heavy roadworks machinery. To this end, simple marking (signage) of roadsides (using colour-coded markers on existing road reflector posts) has been successfully implemented in many council areas to warn road workers of sensitive vegetation areas.

In Australia and elsewhere, it is critical for state and federal environmental agencies to provide necessary resources to local government road managers to conserve roadsides.

Given that conservation outcomes derived from roadsides can greatly contribute to catchment or state based conservation targets, these assets cannot be ignored. Also in terms of future climate change scenarios, a green network is already in place to assist native species to disperse across the landscape. In this context, it is vital that roadside vegetation networks are maintained and even improved with further restoration programs. Ongoing monitoring of roadsides is also vital, as vegetation conditions can both improve and deteriorate, depending on prevailing disturbances (i.e. stock grazing, fire, flood, or soil disturbances from road maintenance activities).

Opportunities for conservation of minor roads – identifying heritage values

The retention of native vegetation along minor roads has many ecological, as well as important economic, aesthetic and social benefits (Breckwoldt 1990). Economic benefits that minor roads provide include a reduction in road management costs associated with conservation priorities, or a rise in ecological-based tourism (Durrant 1994). Social benefits which may occur include the preservation of cultural and natural heritage. Studies have also described important aesthetic benefits for road users travelling along natural landscapes (Cackowski and Nasar 2003). Minor roads also provide service corridors for powerlines, water, sewage, gas, telecommunication and other utilities (Table 1).

An example of one the ‘new’ values that minor rural roads possess is that of an historic road. As discussed, many minor roads in Australia follow the tracks of early explorers and settlers, some of which may have indigenous origins, and therefore possess important cultural heritage values as an historic road. As such – many have a story to tell, which could be a useful approach in interpreting and educating the general public of associated biodiversity conservation values of minor roads. Roads can have important cultural heritage values for the physical structures they possess (such as old bridges), but also for possessing uncommon, rare or endangered aspects of a regions cultural or natural history (e.g rare or endangered plants or ecosystem). In turn, these species, habitats, and remnant ecosystems provide important aesthetic and social values, which are often unique to a local community.

Formal listing of specific road reserves, or stock routes, on state and national registers may result in opportunities to gain additional funding for management from sources otherwise not considered. However the criteria for listing an historic road are often not well understood. The term immediately evokes thoughts of famous roads such as Route 66 in the USA, or the Great Ocean Road in Victoria, Australia. Depending on the legislation that applies, a road can be listed as an ‘historic road’ if it is important in the course, or pattern, of the states cultural or natural history, or possesses uncommon, rare or endangered aspects of the states cultural or natural history (*NSW Heritage Act* 1977, Amended 1998). The benefits of identifying, preserving and managing a minor rural road as an historic road are diverse. They may include opportunities for tourism (ecotourism routes, which include historic and natural values) and economic development, and assistance for restoration of historic structures and features such as bridges, survey trees, indigenous camp sites etc. Such an approach may foster community pride associated with a more comprehensive understanding of a local area's cultural and transportation heritage. Importantly, understanding the development history of roads can provide an important tool to gain new awareness of roadside environmental values, and facilitate greater community investment in their ongoing management (Spooner 2007).

Conclusions

Local councils, NGO environmental organisations, naturalist groups, and other larger state-based and federal agencies face ongoing challenges in managing the competing values of minor rural road networks. In terms of their environmental values, key threats such as invasive species and pollution from adjacent areas require constant vigilance. As human constructions, the key to success in ensuring the persistence of roadside vegetation is in addressing ongoing human impacts. Therefore it is vital that road managers place a greater emphasis on the conservation importance of roadside habitats, and minimise the impacts of disturbances associated with road maintenance and construction activities.

Given the conservation importance of many roadside environments, councils and state managers should "think outside the box" in regards to future road upgrade programs. As described, minor roads often provide refuge for threatened species and ecosystems. Therefore any attempt to upgrade and widen a road will no doubt require expensive mitigation or restoration strategies where conservation values are impacted upon by the development. Alternative strategies need to be explored such as: (1) expanding the road corridor width, by acquiring adjacent farmlands if necessary, to provide opportunities to encourage natural regeneration of native species into surrounding areas, and (2) moving the road – rather than widen the road and impact upon roadside vegetation, it may be feasible to close the road, and build a new carriageway on adjacent cleared farming lands instead. In this way, remnant habitats are left intact.

Linear features such as minor roads are often critical for conserving flora and fauna in rural landscapes, and in providing potential 'conduits' for improving connectivity between fragmented populations (Spooner and Smallbone 2009). The importance of vegetated minor roads will only become more pressing with climate change, where the value in preserving and maintaining vegetated road reserves may be fully realised.

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Abundance of red-listed species in infrastructure habitats – "responsibility species" as a priority-setting tool for transportation agencies´ conservation action

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Abstract

Road and railroad verges may contribute to nature conservation by providing habitat for many species, but due to limited resources, there is a need to select the most important road and railroad stretches for adapted management. We explore the responsibility species concept as a tool for the Swedish Transport Administration to make this selection. We propose lists of candidate responsibility species based on relative abundance of conservation priority species in the vicinity of roads and railroads, respectively. Abundance data were derived from crowd-sourced species observations. Species with ≥20% of observations in infrastructure habitats were included as candidate responsibility species. For roads 32 species were included in the list, for railroads seven species, with an overlap of three species between the lists. We analyzed habitat and management requirements of the listed species to try identifying functional groups. Most of the species require open or semi-open habitats, mainly dry grassland or heathland on sandy or limy soil, un-sprayed crop fields, or solitary trees. Host plants or substrates include broom (genus *Genista*), patches of bare soil, and sun exposed wood. Conservation actions prescribed for the species include, e.g., late or irregular mowing, removal of the field layer, planting of host species, protecting and providing particular substrates, and special protection of certain sites. We argue that road and railroad managers are particularly well suited to conduct most of these actions. We consider the responsibility species concept to be a useful tool for transportation agencies to set priorities for adapted verge management, and the current method to be effective in identifying a first list of candidate species. We discuss the possibility of also identifying responsibility habitats or general management measures based on the results.

Keywords
Infrastructure habitats, Railroad verge, Responsibility species, Road verge, Verge management

Introduction

While the transportation infrastructure has many negative impacts on wildlife, such as habitat loss, disturbance, pollution, mortality and barrier to movements, road and railroad verges may provide important habitat for many species (Way 1977, Forman et al. 2003, Huijser and Clevenger 2006). The sun-exposed grasslands or large trees in roadsides create refuge for many vascular plants and invertebrates that are otherwise in decline due to drastic changes in land use (Eversham and Telfer 1994, Persson 1995, Thomas et al. 2002, Saarinen et al. 2005, Hopwood 2008, Lennartsson 2010). Similarly, railroad switchyards and embankments provide dry, open land often with a diverse flora and invertebrate fauna (Stenmark 2010). Transportation corridors offer a variety of substrates, soil-types and nutrient levels on a small scale, and in agricultural regions, urban areas, and other highly modified landscapes, road and railroad verges may be the only semi-natural habitat that remain (Thomas et al. 2002, Forman et al. 2003, Huijser and Clevenger 2006). Moreover, roads and railroads stretch along the landscape, with a potential to connect remnant habitat patches (Vermeulen 1994, Eversham and Telfer 1994, Tikka et al. 2001, Viles and Rosier 2001, Thomas et al. 2002, Hopwood 2008). As being present in most landscapes, the green network formed by semi-natural strips along roads and railroads has been labelled "a centerpiece of conservation" (Forman et al. 2003).

In Sweden, a large proportion of the state owned road verges has been surveyed for plant biodiversity and some particularly species-rich verges are subject to adapted management, mainly late mowing (Swedish Road Administration 2004). Such species-rich road verges are estimated to cover six per cent of all state owned roads, or a total of 36,000 km, but a long-term goal is to reach ten per cent (Stenmark 2012). Also, some railroad environments, tree rows and large solitary trees along roads have been surveyed (e.g. von Platen 1996, Larsson and Knöppel 2009) and active management to maintain biodiversity has started at some places. Well-designed action for managing species-rich road and railroad verges is essential to keep or enhance habitat quality and to avoid damage to biodiversity during road or railroad maintenance (e.g. mowing, ditching, or spraying) and upgrading (Thomas et al. 2002, Forman et al. 2003). The financial and human resources for such action are however limited, and the Swedish Transport Administration (STA) has asked for means to identify priority road and railroad verges for adapted management.

Here, we explore the possibility of applying the concept of responsibility species (e.g. Dunn et al. 1999, Schmeller et al. 2008) as a priority-setting tool for the transportation agency´s conservation action. A responsibility species is broadly defined as a species for which a large proportion of its entire range or population occurs within the geographic area of an administrative entity (a country, a regional authority etc.). The approach of assigning conservation responsibilities developed in the 1990s, triggered by the shared common responsibility for the protection of global biodiversity agreed in the Convention on Biological Diversity (CBD). National responsibilities for the global conservation of species and habitats is now a developed method for determining conservation priorities, particularly in Europe (Schmeller et al. 2008). Some European countries have also distributed their conservation efforts nationally by assigning responsibility species to county administrations and municipalities (Reck et al. 1996, Larsson 2006, Jooss et al. 2009). However, the concept of responsibility species has to our knowledge not been applied to spatially less coherent administrative units, such as a network of linear infrastructures. Such an approach would support the sectorial involvement in biodiversity conservation required by CBD (article 6b), and we believe that the approach may be fruitful particularly in the case of a national transportation network, where the actors are relatively few and the management decisions centralized.

Over the years, a number of different methods have been used to identify responsibilities for species conservation (Schmeller et al. 2008). Most of these methods combine the species´ conservation status (i.e. red-listing, rarity, or population trend) with a measure of the relative importance of the area in question, based on either distribution or abundance. When dealing with a spatially non-coherent administrative unit, such as a road or railroad network, species distribution can obviously not be used as a basis, due to the coarse geographical level on which distributions are typically defined. Accordingly we used the abundance of a species to investigate the importance of road and railroad sides, and did this for a selection of red-listed species. In order to get the large number of data points needed for this approach, we used databases for crowd-sourced species observations. We propose one method to identify candidate responsibility species for Swedish roads and railroads respectively, and try identifying functional groups regarding habitat and management requirements for the species derived with this method. We discuss the applicability of the responsibility species concept to infrastructure habitats, and the possibility also to identify responsibility habitats or general management measures based on the functional grouping.

Methods

In this quest for responsibility species for the STA, we included only red-listed species subject to a national Swedish initiative for making species-wise action plans (Gärdenfors 2003). Swedish red-listed species have been assigned following IUCN criteria (Gärdenfors 2000), and a sub selection of species for action plans has been done, independently from the present study, by the Swedish Species Information Centre, based on a combination of extinction risk, international responsibility, knowledge level, and susceptibility to management action (Gärdenfors 2003). Hence these species have been identified as priority species by Swedish conservation authorities. Species with action plans also tend to be subject to a larger public interest and therefore better searched for. We excluded species groups that were assumed not to depend on road or railroad verges (fish, mollusks and algae) and species groups that should not be favored near traffic due to the road kill hazard (birds, mammals, reptiles and amphibians).

We derived observational data from the years 1980-2010 from two data bases available within an internet-based platform for voluntary reporting of species observations: the Swedish species observation data base and the Species Gateway (http://www.artportalen.se), both managed by the Swedish Species Information Centre. The two reporting systems are not overlapping with regard to observations and were therefore treated as one dataset. Each observation in these bases contains data on location (with estimated accuracy), date and observer as a minimum; sometimes additional data such as habitat type are given. Only observations with a reported precision <100 m were included. Observations from the same geographical point were reduced to one observation, to exclude double counts.

Of the remaining observations, those within 50 m from state owned roads or railroads (measured from the road or railroad center) were attributed to the infrastructure habitat. This distance was arbitrarily set to cover the entire road or railroad corridor. We are aware that this procedure resulted in some observations outside of the actual corridor being attributed to the infrastructure habitat. We do not see this as a problem however, because it can be argued that species dwelling in such proximity to a road or a railroad may be affected by its management, and may therefore qualify as a responsibility to the infrastructure manager.

Only species with a minimum of three observations were included in the analyses, to avoid the most obvious randomness in results. We calculated the proportion of all observations of each species in infrastructure habitats in relation to the country as a whole. As a cut-off value for inclusion in the list of candidate responsibility species we set 20%, because it is roughly in line with what has been previously applied to assign responsibility species in Sweden (Larsson 2006) as well as other Scandinavian countries (e.g. Stoltze and Pihl 1998). The Swedish road and railroad network covers roughly 1.5% of the total land surface (Seiler and Folkeson 2006), so by selecting 20% as cut-off, we hope to have included only species that are disproportionally often found in infrastructure habitats. Road sides and railroad sides were analysed separately. In order to identify functional groups among the candidate responsibility species, we summarized habitat requirements and management action proposed in the action plans and in species fact sheets linked to the red-list (references given in Table 1–2).

Results

Of the total of 308 species included in the analysis, 197 species (64%) were reported with at least one observation in road habitat and 53 (17%) with at least one in railroad habitat. The lists of candidate responsibility species (i.e. ≥20% of observations in infrastructure habitat) contained 32 species for road habitat (Table 1) and seven for railroad habitat (Table 2). The overlap between species in road and railroad habitats was large; 47 of the 53 species found in railroad habitat were also found in road habitat. Among candidate responsibility species, there was an overlap of three species: the longhorn beetle *Phytoecia nigricornis*, the bee *Anthophora plagiata*, and the leaf mining moth *Phyllonorycter staintoniella*. The total number of candidate responsibility species derived with this method was therefore 36. Among these species, 23 were invertebrates (all seven in railroad habitat), seven vascular plants, and six fungi/lichens (Table 1–2). Among the invertebrate species, *Lepidoptera* (moths and butterflies) were the main taxonomic group, with 12 species.

Almost all of the listed species require some kind of open habitat, such as grassland, heathland, crop field, or a general openness in the surroundings (holds also for species on trees, fence poles and buildings). Nine species on the road list grow in managed grasslands and three in un-sprayed crop fields. Dry or sandy soil (or both) is required by 14 and six species on the road and railroad lists, respectively; limy soil is required by five species on the road list. Ten road species and one railroad species are favored by bare soil during at least part of their life cycle, either through physical disturbance of the field layer (such as erosion, livestock tramping, or driving with vehicles) or in crop fields. Five of the seven species on the railroad list live on broom (genus *Genista*). Large or solitary trees create habitat for seven species on the road list, and processed timber in fence posts or wooden houses is substrate for four species on the same list. A particular preference for warm microclimate (sun exposed sites, such as southern slopes or wood in solitary trees) is described for eight of the species on the road list and four on the railroad list.

In accordance with the species´ requirements described, clearing of shrub in order to keep habitat openness is a common theme in the prescribed management. Many of the species are in need of maintained or adapted (late or irregular) mowing, sometimes in combination with field layer removal (soil scarification or burning). Species growing in crop fields require adapted farming (choice of crop and seed mix, no chemicals, etc.). In some cases, protecting and providing particular substrates (hollow trees, dead branches, unproofed timber) or planting host species is proposed. For species where the remaining sites are particularly few, special protection of these sites, and fine-tuned management in consultation with local conservation authorities, is advised. Sustained or strengthened population monitoring is recommended for all species (therefore not mentioned in Tables 1–2), to keep track of population trends and effects of conservation measures.

Discussion

The importance of certain road and railroad verges as habitat refuges for rare or declining species has been reported from previous research and conservation case studies, both in Sweden (Persson 1995, Weidow 2008, 2009, Stenmark 2010, Lennartsson 2010) and other countries (the Netherlands: Vermeulen 1993, Eversham and Telfer 1994, Van Rossum et al. 2004, Schaffers et al. 2012; Belgium: Tanghe and Godefroid 2000; Great Britain: Thomas et al. 2002; Finland: Koivula et al. 2005; South Africa:

Table 1. Priority species for Swedish conservation (=red-listed species with species-wise action plans) with at least 20% of the observations nationally in road habitats. Habitat requirements and management described derive from general descriptions, and are not particularly adapted to road habitats. Species in bold are found also in Table 2.

Species	Total no of obs.	% in road-sides	Habitat requirements (substrate, host species, microclimate etc.)	Management requirements	Source
Phytoecia nigricornis (a longhorn beetle)	3	66.7	Dry, sandy soil with *Solidago virgaurea*, sun exposure	Clearing of shrub	1
Sphinctrina anglica (a lichen)	13	61.5	Fence posts, timber houses	Protecting current substrates and providing new	2
Genista germanica (German greenweed)	10	60.0	Dry heaths, sun exposure	Soil scarification, burning, clearing of shrub	3
Andrena labialis (a bee)	31	58.1	Dry, sandy soil with *Fabaceae,* partially bare soil	Adapted mowing, soil scarification	4
Minuartia viscosa (Sticky sandwort)	8	50.0	Dry, sandy, limy soil, sun exposure	Clearing of shrub, soil scarification	5
Senecio erucifolius (Hoary ragwort)	16	50.0	Dry, limy grasslands	Mowing, clearing of shrub	6
Astragalus penduliflorus (Mountain lentil)	125	43.2	Sandy soil	Late mowing, clearing of shrub	7
Eupoecilia sanguisorbana (a moth)	5	40.0	Grasslands with *Sanguisorba officinalis*	Adapted mowing, favouring host species	8
Anthemis cotula (Stinking chamomile)	27	37.0	Edges of un-sprayed crop fields	Adapted farming, no pesticide use, soil scarification	9
Caloplaca furfuracea (a lichen)	47	36.2	Fence posts, timber houses	Protecting current substrates and providing new	2
Rhinanthus serotinus ssp. apterus (a yellow-rattle)	17	35.3	Un-sprayed crop fields	Adapted farming, no pesticide use	9
Andrena batava (a bee)	3	33.3	Solitary willow shrubs, sandy, partially bare soil	Protecting solitary willow, soil scarification	10
Andrena morawitzi (a bee)	3	33.3	Solitary willow shrubs, sandy, partially bare soil	Protecting solitary willow, soil scarification	11
Chlorophorus herbstii (a longhorn beetle)	3	33.3	Tree branches, oak fence posts, sun exposure	Clearing of shrub, keeping dead branches	12
Diploicia canescens (a lichen)	3	33.3	Old, solitary, deciduous trees	Protecting solitary trees and tree rows	13
Agonopterix atomella (a moth)	7	28.6	Dry heaths with *Genista*	Soil scarification, burning, clearing of shrub	3
Coleophora conyzae (a moth)	7	28.6	Limy grasslands with *Inula*, sun exposure	Mowing, clearing of shrub	14
Albatrellus cristatus (a fungus)	8	25.0	Bare, hard soil in beech forest	Protecting beech trees at current sites	15
Anthophora plagiata (a bee)	4	25.0	Bare clay, tiled houses, sun exposure	Protecting current substrates and providing new	16
Phyllonorycter staintoniella (a moth)	4	25.0	Dry heaths with *Genista*	Soil scarification, burning, clearing of shrub	3
Ramaria roellinii (a fungus)	4	25.0	Dry, sandy, limy grasslands with partially bare soil	Soil scarification, clearing of shrub	17
Andrena hattorfiana (a bee)	981	23.6	Dry, sandy grasslands with *Dipsacaceae*	Late mowing, clearing of shrub	18
Eucosma scorzonerana (a moth)	17	23.5	Grasslands with *Scorzonera humilis*	Mowing, clearing of shrub and forest	19

Species	Total no of obs.	% in road-sides	Habitat requirements (substrate, host species, microclimate etc.)	Management requirements	Source
Aethusa cynapium var. agrestis (Fool´s parsley)	44	22.7	Un-sprayed crop fields	Adapted farming, no pesticide use	20
Cyphelium trachylioides (a lichen)	333	22.5	Fence posts	Protecting current substrates and providing new	2
Plebejus argyrognomon (Reverdin's blue butterfly)	76	22.4	Small, sun exposed clearings with *Astragalus glycyphyllos*	Late mowing, planting host species	21
Canthophorus impressus (a true bug)	83	21.7	Dry grasslands with *Thesium alpinum*	Mowing, clearing of shrub	22
Exocentrus adspersus (a longhorn beetle)	33	21.2	Oak branches, sun exposure	Keeping dead branches	23
Lycaena helle (Violet copper butterfly)	105	21.0	Grasslands and springs with *Polygonum viviparum*	Adapted mowing	24
Aphodius merdarius (a dung beetle)	87	20.7	Dry grasslands with horse dung	Mowing, providing new substrate	25
Cheiridium museorum (a pseudoscorpion)	15	20.0	Hollow deciduous tress	Protecting hollow trees, keeping dead wood	26
Triaxomasia caprimulgella (a moth)	5	20.0	Hollow deciduous tress	Protecting hollow trees	27

Sources for descriptions of requirements: 1) Nilsson 2013, 2) Hermansson and Jonsson 2011, 3) Larsson 2007, 4) Karlsson et al. 2011, 5) Mattiasson 2009, 6) Andersson 2006, 7) Grundström 2009, 8) Elmquist 2007, 9) Kloth 2007, 10) Johansson 2012, 11) Johansson 2013, 12) Ehnström 2006, 13) Arup 2006, 14) Lennartsson and Björklund 2014, 15) Svensson and Ryberg 2008, 16) Cederberg 2013a, 17) Nitare 1997, 18) Cederberg 2013b, 19) Björklund and Palmqvist 2007, 20) Aronsson and Jonsell 1996, 21) Elmquist 2010, 22) Karlsson 2011, 23) Franc 2013, 24) Lindeborg 2014, 25) Ljungberg 2007, 26) Sandström 2007, 27) Bengtsson 2011.

Table 2. Priority species for Swedish conservation (=red-listed species with species-wise action plans) with at least 20% of the observations nationally in railroad habitats. Habitat requirements and management described derive from general descriptions, and are not particularly adapted to railroad habitats. Species in bold are found also in Table 1.

Species	Total no of obs.	% in road-sides	Habitat requirements (substrate, host species, microclimate etc.)	Management requirements	Source
Phyllonorycter staintoniella (a moth)	4	50.0	Dry heaths with *Genista*	Soil scarification, burning, clearing of shrub	1
Syncopacma suecicella (a moth)	4	50.0	Dry heaths with *Genista pilosa*, sun exposure	Soil scarification, burning, clearing of shrub	1
Mirificarma lentiginosella (a moth)	13	38.5	Dry heaths with *Genista*	Soil scarification, burning, clearing of shrub	1
Phytoecia nigricornis (a longhorn beetle)	3	33.3	Dry, sandy soil with *Solidago virgaurea*, sun exposure	Clearing of shrub	2
Scythris crypta (a moth)	11	27.3	Dry heaths with *Genista pilosa*, sun exposure	Soil scarification, burning, clearing of shrub	1
Anthophora plagiata (a bee)	4	25.0	Bare clay, tiled houses, sun exposure	Protection of current sites, providing new substrate	3
Coleophora genistae (a moth)	5	20.0	Dry heaths with *Genista pilosa*	Soil scarification, burning, clearing of shrub	1

Sources for descriptions of requirements: 1) Larsson 2007, 2) Nilsson 2013, 3) Cederberg 2013

Tshiguvho et al. 1999; USA: Hopwood 2008). The present study confirms these results, and consolidates the picture by showing that a number of priority species for Swedish conservation actually have a large proportion of their known sites in or near transport infrastructures. The future conservation status of these species can actually be considered to be in the hands of the road and railroad manager to a large extent. Hence, a certain commitment of the transport administration to the conservation of these species appears crucial. Adopting them as responsibility species would be one way to manifest this commitment, and to help prioritizing among conservation actions.

With the current method, a number of candidate responsibility species were identified for Swedish road and railroad management, respectively, based on the national list of action-plan red-listed species. We acknowledge that the method could be applied differently, for example with a different selection of species to be analyzed (such as all red-listed species, or the inclusion of vertebrates), or using another cut-off level than 20% for inclusion. A sensitivity analysis of the cut-off level could further inform a future selection of responsibility species and management action. Also the analysis could be applied on subsections of the infrastructure network in order to identify regional or local priorities. What species to finally adopt is a management decision that should be guided not only by the relative abundance in infrastructure habitats, but also by how well the specific requirements can be met by the infrastructure manager. Another aspect is whether certain actions serve several species; then all these species can be adopted, also those with a lower proportion of occurrences in infrastructure habitats. Hence, supplementing methods may be needed to arrive at a final list of species. The ultimate goal should be to find cost-efficient solutions for species conservation on the national level.

Although we did not analyze the habitat and management requirements of our candidate species systematically, we believe that some patterns of relevance to infrastructure management are still evident. Most of the species require open habitats; mainly dry grassland or heathland on sandy or limy soil, or un-sprayed crop fields. Particular open-habitat requirements include *Genista*-heaths, patches of bare soil, and sun exposed trees and fence posts of un-proofed wood. The requirements indicate that a list of responsibility species will help the infrastructure manager in selecting and adapting actions.

The preference for openness among species in infrastructure habitats is indeed not surprising, but it illustrates well that the clearing of shrub encroaching into road and railroad verges that is done for traffic safety also may have a value for species conservation. The same holds for roadside mowing. If minor adjustments of the present road maintenance operations (regarding extent, frequency, timing, machinery, etc.) are needed to reach the specific management requirements stated in the respective management plan, this may be achieved to a small cost. Similarly, soil scarification could be conducted efficiently with diggers engaged for other purposes in road management. In the case of soil scarification however, care must be taken not to facilitate invasion of alien species.

Host plants for insects, for example native broom species (*Genista* sp.), can be planted or promoted in verges near the present sites of the respective species, or even be integrated in standard landscaping programs. Tree management operations in the road environment already focus on creating tree continuity, with planting of new tree rows, gradual tree replacement, and to an increasing extent leaving old trees and dead wood in place or piled in safe places (Stål and Bengtsson 2010). Also here, minor adjustments to match the needs of particular species may be achieved to a small cost. Species growing in un-sprayed crop fields seem less obvious to promote in association with road maintenance, although soil scarification conducted in distant parts of the verge may be of some help for the survival of local populations.

Preferences of responsibility species can also guide the selection of road or railroad sections or regions to focus on for habitat enhancement. For example, our results suggests that roads and railroads going through areas with sandy or limy soils are likely to have better conditions as habitat for rare species, and for the spreading of individuals from and to the surroundings. Another example is that sun exposed slopes generally provide better conditions. The habitat preferences of responsibility species may be translated into a corresponding list of responsibility habitats; such a list is likely to be an equally efficient tool in infrastructure planning. Regional differences in abundance of the responsibility species may further indicate in what management districts habitat enhancement is of higher priority.

Because of the 50 m buffer from the centerline used to define infrastructure habitat, some of the identified species may indeed not occur in the infrastructure corridor as such, but only by coincidence have their last residences near infrastructure. The current method cannot be used to identify these species, but must be complemented by field visits. For such species, the management of the actual verges may not be critical, but rather that special attention is paid during ordinary road or railroad maintenance and upgrading, so that the sites are not spoiled by mistake. Hence, also here the responsibility species concept can potentially aid the choice of action.

When providing habitat for animals in the vicinity of roads and railroads, clearly the risk of creating ecological traps (e.g. Schlaepfer et al. 2002) must be considered. Amphibians and reptiles as well as mid-sized and large mammals can experience high levels of traffic mortality (Fahrig and Rytwiski 2009), and accordingly we have not included vertebrates in our study. Concerning insects however, available research suggests that traffic mortality rarely has effect on the population level (Munguira and Thomas 1992, Hopwood 2008, but see Weidemann and Reich 1995), and that the benefits from infrastructure habitats outweigh the hazard from passing vehicles (Thomas et al. 2002, Hopwood 2013, Skorka et al. 2013), but the matter deserves further study. We suggest that the hazard will be minimized by directing habitat improvement or creation to low traffic infrastructure, or to the parts of the infrastructure corridor that is most distant from the traffic.

One management action required for all the listed species is keeping track of the population development, both in and outside infrastructure habitats. Data on how the relative importance of infrastructure habitats develops with time may justify revision of the responsibility species list, and also give insight in the success of measures taken to conserve the species in different habitats.

Our results illustrate the importance of infrastructure habitats in relation to other habitats in the landscape, and hence points at the necessity for transport administrations to be integrated in nature conservation on the landscape scale. With “the centerpiece of conservation” (sensu Forman et al. 2003) in their hands, infrastructure managers may even function as catalysts for cross-administrational conservation efforts. Effective partnerships are essential. A conservation responsibility should not be misinterpreted as a duty for a single party, or an opportunity for other administrative entities to resign.

In the present case, crowd-sourced species observation appeared to be a useful data source. Despite the obvious drawback of lack of systematic sampling design in this type of data collection, the extensive spatial coverage and large number of observations still make them an interesting potential (Snäll et al. 2011). One limitation in the present study is that the method chosen to identify what may be labeled ”infrastructure species” only took state owned roads into consideration. Privately owned roads add up to more than 75% of all roads in Sweden (Swedish Transport Administration 2014), most of them small, unpaved, and well dispersed in the landscape. Hence, some species not identified as candidate species for the STA´s responsibility may well depend on the management of minor roads (Tikka et al. 2000).

Conclusions

We consider the responsibility species concept to be a potentially useful tool for setting priorities for the STA´s action to contribute in the conservation of endangered species. The described method could be used to point out a number of candidate responsibility species, and to outline important management actions. The most immediate action needs to be directed to and near the remaining sites of a limited number of species. In the longer term, the specific habitat and management requirements of responsibility species may help indicating road or railroad stretches or regions where an adapted management can effectively create new habitat for threatened infrastructure species. Put in a larger perspective, our study points out the crucial role that transportation administrations may have in landscape-scale nature conservation.

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